The detection of biotic changes in the Tekapo riverbed after habitat restoration

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dedicated to mum and Mf

Abstract

The impact of bulldozing on terrestrial invertebrates in the Mackenzie Basin (South Canterbury) was investigated. Willow trees (*Salix* spp) had been bulldozed from the Tekapo delta to recreate braided riverbed habitat for riverine birds. Poor survival of these birds is attributed to predation by cats and ferrets. Predators supplement their predominantly rabbit diet with birds, lizards and invertebrates when rabbits are scarce. An increase in predation pressure on riverine birds could result if invertebrate prey decreased in availability following bulldozing. Invertebrates were sampled with pitfall traps across the Ohau and Tekapo deltas, before and after willow clearance. Eighteen sites represented different time periods since willow clearance. A comprehensive vegetation-environmental variable survey was also made. Data were analysed with the simple statistical measures of species richness, evenness and diversity, and the more complex methods of cluster analysis and ordination to determine if distinct invertebrate and plant communities existed. Simple statistics were inadequate to distinguish between 'Shingle', 'Willow', 'bulldozed/disturbed' and 'braided riverbed' habitats. Community classification and ordination techniques were more satisfactory. The inclusion of plant species data in the invertebrate ordination accounted for twice the variation explained by the macro-environmental data and by the invertebrate ordination alone. Improvement in habitat for riverine birds was measured by the appearance of braided riverbed taxa in the bulldozed areas. Very few 'braided riverbed' invertebrate or plant taxa appeared in the bulldozed areas even two years after willow clearance. The invertebrate prey of mammalian predators were predominantly weta but these contributed little to the diet by weight. Seven undescribed species of invertebrate were identified as having conservation value. These included a new species of *Hemiandrus* (a ground weta), and *Prodontria* (a chafer beetle).

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Figure 4.12.2. *Prodontria* new sp. (right) next to the common grass grub beetle *Costelytra zealandica* (left). (actual size about 1.5 cm long). Photo courtesy of Ed Walls.

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Chapter 1
Introduction

Background to this study.

The Braided riverbeds of the Mackenzie Basin provide feeding and breeding grounds for several birds especially Charadriiformes (Pierce, 1979; Pierce, 1983). Of the six bird species identified by Hughey (1985) as having braided rivers as their preferred habitat, three were considered to be vulnerable to extinction (vis. Banded Dotterel *(Charadrius bicinctus)*, Black fronted Tern (*Sterna albostriata*) and Wrybill (*Anarhynchus frontalis*) and one was endangered (the Black Stilt *Himantopus novaezealandiae*). Black Stilt populations have been actively managed since 1984 through a captive breeding program by the NZ Wildlife Service and Department of Conservation. Wild breeding pairs of Black Stilts are only found in the braided riverbeds of the Mackenzie Basin. It is one of the world's rarest wading birds and is classified in category 'A' (Highest conservation priority requiring management in the short term) in the conservation hierarchy set out by Molloy and Davis (1992) and subsequently Tisdall, (1994). Wild populations are being enhanced through rearing of eggs and release of juveniles (Reed and Murray, 1993).

The majority of riverine birds nest and roost on areas of bare shingle with sparse vegetation (typical of early stage succession within a braided riverbed). Nest site preferences range from the predominantly bare yet flood prone areas used by Black-billed Gulls (*Larus bulleri*) and Wrybills to the stable shingle partially covered sites with mat plants (*Raoulia* spp), *Epilobium melanocaulon* and *Muehlenbeckia axillaris* used by Black fronted Terns and Banded Dotterels (Soper, 1959; Pierce, 1983; Bomford, 1986). The Banded Dotterel, Wrybill, Black fronted Tern and Black Stilt are territorial and defend nesting sites. Maintenance of self sustaining populations is critically dependent on preservation of breeding grounds in braided habitat (Hughey, 1990).

Many of the Mackenzie Basin rivers (and lakes) have been modified through extraction and diversion of water for hydro-electric power generation. (Of the main tributaries to the Waitaki system, only the Ahuriri and the Cass River remain largely

unmodified). The majority of water from the main rivers has been diverted down canals through power generation stations to such an extent that the mean flows are significantly lower than natural flows, and duration and timing of flood events (which create and maintain the braided river state) have been affected, and large expanses of riverbed have been flooded to create impoundments. This has resulted in a loss of suitable nesting, breeding and feeding habitats for many riverine birds (Hughey, 1990).

The majority of these bird species (e.g. Black Stilt, Black fronted Tern) feed on aquatic invertebrates and fish (Lalas, 1979; Robertson and Dennison, 1979; Robertson *et al*, 1984; Pierce, 1982; Pierce, 1986). Some species (e.g. Wrybill, and Banded Dotterel *(Charadrius bicinctus)*) exhibit diet switching and rely heavily on terrestrial invertebrate prey when the abundance of river-insects is low after flood events. Pierce (1979) observed that Wrybills switched from a relatively narrow (stenophagous) diet to a broad (euryphagous) diet which included riparian invertebrates such as Carabidae, Araneae and very small flies (probably Simuliidae).

Reduction in flood events has allowed the invasion of exotic weed plants such as crack willow (*Salix fragilis*), lupin (*Lupinus* spp.), sweet briar (*Rosa rubiginosa*) and pasture grasses and herbs (e.g. browntop *Agrostis capillaris* and white clover *Trifolium repens*) and many rivers have become channelised as a consequence (Hughey, 1990), which results in scouring of shallow feeding areas. Species like the Wrybill, Dotterel and Black Stilt are territorial and defend areas of habitat and therefore the amount and quality of the habitat is important in contributing to high bird densities and their survival. Permanent regeneration of vegetation is one of the major long-term threats to the stability of populations of Wrybills, Black fronted Terns, Banded Dotterels and Black Stilts (Pierce, 1983; Marchant and Higgins, 1993). Hughey (1982) found that moderate to dense vegetation growth restricted the availability of potential nest sites, while Robertson *et al.,* (1984) noted that an increase in the stability of the Ahuriri catchment from weed invasion led to a decline in bird density and a change in species composition through competition for breeding sites (Wrybill nesting decreased following an increase in vegetation-tolerant Black-backed Gulls (*Larus dominicanus*) (Palmer, 1990).

Maintenance of self sustaining riverine bird populations is critically dependent on preservation of braided (tree and shrub-free) habitat (Hughey, 1990). Existing wildlife values of many of the affected areas is low but some rivers have much potential if their habitat is improved by making them tree and shrub-free (Hughey, 1990).

Project River Recovery.

Project River Recovery is a compensatory agreement made in 1990 between Department of Conservation and the Electricity Corporation of New Zealand (ECNZ) in recognition of the impact of hydroelectric development on wildlife habitat. Its prime objective is to restore or enhance wetland habitat (for birds) in the Upper Waitaki, equivalent to or greater than the net loss of habitat and conditions attributable to Waitaki hydro-electric power development (Warren, 1994). Funding for the Project is linked to the rights of ECNZ to use the water resources of the Waitaki catchment and is guaranteed for the duration of the water rights (up to 35 years) and is to be revised every seven years (Rawlings, 1993; Taylor, 1993). Objectives of Project River Recovery include the control of introduced vegetation in the Ahuriri and lower Tekapo Rivers, the creation or modification of shallow pond wetland systems for wading birds particularly the Black Stilt, and contribution to the Black Stilt rearing programme for release of more birds into the wild (Warren, 1994). Several research projects (including the present study) are also subsidised from Project River Recovery.

The control of introduced vegetation includes the removal of Willow trees (*Salix fragilis*) and shrubs from the lower Tekapo riverbed below the confluence of the Tekapo and Pukaki Rivers with the aim of creating a braided riverbed-type habitat suitable for riverine birds. The nearby Ohau river is as close to this point as is possible given the current level of management, and it is used as a nesting habitat by several riverine birds. The actual clearance work was contracted out to the Canterbury Regional Council (who also maintain flood channels) (P.R.R Green pamphlet). Large willow trees were lifted out with diggers, piled into windrows to dry, burned and the remainder buried. Control of willow regrowth and other weeds was carried out by applications of herbicides. The lower portion of the Tekapo delta

(from lake Benmore up to the pylon line and from the terrace on the true right to Haldon camp on the true left side) was cleared in 1992; the remainder was cleared in 1993.

Predators.

Reduction in the number and size of braids has also allowed mammalian predators greater access to the (previously safe) riverbed islands and contributed to a major increase of predation pressure on riverine birds (pers. comm., Richard Maloney, Twizel Conservancy). Predators of birds are predominantly introduced mammals *vis.* feral cats (*Felis catus*), rats (*Rattus* spp), and Mustelids (*Mustela furo* and *M. erminea*) (Pierce, 1987). Predators are known to have profound impacts on native bird fauna, especially on islands. Fitzgerald and Veitch (1985) recorded a loss of six bird species from Herekopare Island since the introduction of cats. Dilks (1979) noticed a decline of burrowing petrels on Campbell Island as a result of cat predation. Cats seriously affected the breeding success of the Black Petrel (*Procellaria parkinsoni*) and the extinction of the Saddleback (*Philesturnus carunculatus*) on Little Barrier Island. Dramatic video footage has captured evidence of Dotterel nest predation by a cat in the Mackenzie basin (P.R.R conference, Twizel, 1993). Predation was cited as one of the major causes contributing to the decline in Black Stilt populations (Pierce, 1982).

However, where mammalian prey (lagomorphs and rodents) are available, these comprise the major proportion of predator diet (Fitzgerald and Karl, 1979; Catling, 1988; Fitzgerald *et al.*, 1991; Murphy and Dowding, 1994). Ferrets and cats in the Mackenzie basin prey mainly on rabbit (*Oryctolagus cuniculus*). Rabbit are a major pest in the Mackenzie basin, and populations are periodically poisoned to control them (Pierce, 1987). Predators may exhibit a functional response to low mammalian prey density by diet switching (e.g. Fitzgerald and Karl, 1979; Konecny, 1987; Pierce, 1987; Murphey and Bradfield, 1992). Alternative prey includes lizards, birds (native and introduced) and invertebrates (Fitzgerald and Veitch, 1985).

Predators in the Mackenzie basin are known to supplement their diet (Pierce, 1987, Pascoe, 1995). Pierce (1987) found that female ferrets and juvenile cats

supplemented their predominantly rabbit diet with small vertebrates (birds and lizards) and invertebrates (mostly weta). A decrease in rabbit availability was followed by increased predation pressure on nesting birds in the next breeding season, and stoats immigrated following a decline in ferret and cat abundance. During the recovery of prey populations, rabbits which were actively hunted provided a buffer to bird predation. If the availability of the invertebrate prey changes as a result of habitat creation, then changes of predation pressure on the riverine birds could result which may hinder conservation efforts.

Invertebrates and Conservation.

In 1984, Espie *et al.*, surveyed the Mackenzie Ecological Region for inclusion in the Protected Natural Areas programme. The programme is designed to preserve representative samples of natural ecosystems which give New Zealand its recognisable character. Although their survey emphasised the conservation of plant communities, it did to a very limited degree sample and list some of the invertebrates from the recommended areas (Patrick, 1992).

Until recently, conservation of invertebrates has been overlooked on account of their small size and relative invisibility (Crawley, 1982). Conservation of New Zealand invertebrates has focused on conspicuous single taxa such as weta (e.g. Gibbs and Richards in Cresswell and Veitch, 1994; Townsend, 1995), Land snails (Sherley, 1994), weevils (Emberson, 1995; Schops, 1995) and other large invertebrates (Meads, 1990). Present invertebrate management techniques include legislation (selected invertebrates were included in the Wildlife Amendment Act in 1980 (Crawley, 1982)), reserves (e.g. Cromwell chafer reserve for *Prodontria lewisi* (Watt, 1979)), island transfers (transfer of the giant weta *Deinacrida rugosa* from Stephens Island to Maud Island; (Timmins *et al.*, 1987; Meads and Notman, 1992)) and captive rearing (e.g. Mahoenui giant weta (Dugdale, in Cresswell and Veitch 1994)).

Two grasshoppers (Orthoptera) from the Mackensie Basin already have recognised conservation value. These are *Brachaspis robustus*, and *Sigaus minutus*. *Brachaspis* was moved from category I (species about which little information exists,

but which are considered to be threatened) in 1992 to category A (urgency for action, highest priority threatened species) in 1994 in their ranking system for endangered species (Molloy and Davis, 1992; Tisdall, 1994). It inhabits stony areas, but due to the stabilization of the riverbeds through hydroelectric development and encroachment of trees onto the riverbeds, the habitat of this species has been severely reduced. Populations are now confined to stony areas within the Mackenzie Basin (White, 1994).

Other invertebrates of significant value may exist in this area. However, very little is known about the invertebrate faunal communities from the Mackenzie Basin. Although the habitat restoration aims to benefit riverine birds, little is known of the impact this will have on invertebrates. It is desirable to minimise the impact on important invertebrate taxa such as rare natives (e.g. *Brachaspis robustus*), or undescribed species.

Very few studies exist on communities of New Zealand terrestrial invertebrates, partly because of the paucity of information on New Zealand invertebrates in general. Literature on Mackenzie Basin area mainly concerns changes and deterioration of tussock grassland (Dick, 1940; Connor, 1964; Hubbard and Bastow Wilson, 1988; Treskonova, 1991), but corresponding studies on the impact of this on terrestrial invertebrates are few (e.g. Kelsey, 1957; Harrison and White in Knox, 1969; White, 1991). Similarly, little is known about riverbed invertebrate fauna. Riverbed vegetation in general has been widely documented (e.g. Calder, 1961; Reed in Knox, 1969; Burrows, 1977). Wardle (1991) observed that excepting urban wasteland, riverbeds are one of the richest habitats for naturalised plants. However, studies on terrestrial riverbed invertebrates mainly concern individual taxa (e.g. the genus *Bembidion* (Lindroth, 1976) and native bees (Quinn, 1984)).

The effects of disturbance on stream invertebrates and waterfowl has received profuse attention (e.g. Hughey, 1990; Weatherley and Ormerod, 1990) but the impact on terrestrial invertebral communities is relatively unknown. It is therefore not surprising that no literature exists concerning the impact of bulldozing during habitat creation for birds (within Project River Recovery) on Mackenzie Basin invertebrates.

The impact of this habitat disturbance must therefore be inferred from related fields of research. These include the effect of land management on soil fauna (Yeates, 1991), the effects of herbicide on soil invertebrates (Malinda *et al.*, 1981), recolonisation by grasshoppers of reclaimed strip mines (Parmenter *et al.*, 1991), changes in grasshopper assemblages in rangeland (Fielding and Brusven, 1993), effects on invertebrates of conservation management in reedbeds (Dilthogo *et al.*, 1992) responses of Hemiptera and Heteroptera to cutting of grassland (Morris, 1979; Morris and Lakani, 1979) and effect of heathland fragmentation on invertebrates (Webb and Hopkins, 1984; Hopkins and Webb, 1984; Webb, 1988), and the effects of volcanic activity on White Island invertebrates (Hutcheson, 1992).

Several trapping methods exist to investigate invertebrate communities (e.g. Southwood, 1978), and there are many ways of summarising community data (e.g. Jongman *et al.*, 1987; Magurran, 1988). The detection of an impact from bulldozing on invertebrate communities will vary depending on the trap technique chosen and the statistical treatment of the data. Appendices 2 and 3 provide some insights into problems of data collection and analysis.

Objectives.

1) To compile a basic list of invertebrates and plants that occur in the Tekapo and Ohau River Deltas.

2) To investigate plant and invertebrate assemblages at site, habitat and community level using simple and complex statistics to determine which methods are most useful in distinguishing between 'willow', 'shingle', 'braided riverbed' and 'bulldozed/disturbed' habitats, and to investigate which methods are the best predictors of invertebrate habitat for conservation purposes.

3) To investigate whether habitat quality for riverine birds has improved after bulldozing, measured by the appearance of 'braided riverbed' taxa (plants and invertebrates) in bulldozed habitats one and two years after clearance.

4) To investigate the impact of bulldozing on invertebrates which are consumed by predators so that the likelihood of a change in predation pressure on riverine birds can be assessed. (This objective will be carried out in conjunction with other researchers).

5) to investigate the presence of invertebrate taxa which may have conservation value (i.e. by being rare native or endemic species, undescribed species, or have localised or restricted habitats and distribution).

Chapter 2
Study site

History and vegetation.

Nearly all of the Mackenzie Country is grassland. Natural and Polynesian fires depleted Beech and Podocarp forest and removed forest from the central basin (O'Connor, 1976). Sheep were introduced into the Mackenzie Basin in 1857 (Treskonova, 1991). Early pastoralists used fire to open up the tall tussock (*Chionochloa* spp) which was replaced by short tussock (*Festuca* spp) (Connor, 1964). European burning and grazing greatly extended the range of short tussock grassland at the expense of tall tussock.

Serious degradation of the short tussock grassland accompanied rabbit infestations and overgrazing by sheep in times of financial stress (often also accompanied by drought e.g.1929-1934 (O'Connor, 1976)). This allowed adventive species introduced as contaminants in seed stock to invade intertussock spaces leading the predominance of rosette forming species (e.g. *Hieracium* spp.), and other naturalised adventives such as Sorrel (*Rumex acetosella*), *Hypericum perforatum*, *Echium vulgare* (Williams, 1980; Treskonova, 1991) and sweet brier (*Rosa rubiginosa*). Short tussock grasslands were first acknowledged in the 1920s to be induced from human modifications rather than climate. Recovery rates are slow in this dryland area (Espie *et al*, 1984). " Willows now characterise margins of the main river channels and broom has invaded the riparian lands of the lower montane and lowland zones of the basin " (O'Connor 1976; p 8).

Climate.

The basin receives low mean annual rainfall (< 500 mm) and is considered semi-arid or sub humid and experiences the driest conditions and most extreme temperatures recorded in New Zealand (Connor, 1964; Williams, 1980; Espie *et al*, 1984) with very warm summers and cold winters.

Geology.

The Mackenzie Basin is bounded to the north and west by the Main Divide,

to the east by the Two Thumbs range, Rollesby, Kirkliston and Dalgety ranges and to the south by the Ohau and Waitaki rivers. Extensive moraines and outwash plains dissected by rivers and streams are characteristic of the basin. The basin around the Tekapo river has Pliocene fluvioglacial gravels and sands overlying tertiary sediments. Outwash plains are of well graded Greywacke gravel (Connor, 1964). Soils are dry subhygrous to hygrous high country yellow-brown earths with low fertility, and are prone to sulphur deficiency. They are strongly leached despite the low mean annual rainfall (Espie *et al*, 1984).

Hydro-electric development.

The Tekapo, Pukaki and Ohau rivers once drained the natural moraine-dammed lakes of Tekapo, Pukaki and Ohau respectively. Large areas have been modified for hydro-electric power generation. Installation and manipulation of the lakes has altered the seasonal distribution of water flows in the Waitaki basin. Extra storage capacity has meant that many of the arid areas now hold large bodies of static water all year round. Downstream flow patterns are now more constant throughout the year with fewer (and lesser) fluctuations (Robertson *et al.*, 1984). Mean flows are significantly lower than natural flows and the intensity, timing, duration and frequency of flood events which contribute to the braided riverbed state have been affected.

The majority of water which would otherwise flow into the Tekapo, Pukaki and Ohau rivers has been diverted into canals which supply hydro-electric generating power stations. Lake Pukaki control structure was completed in the 1950s. In 1954, the Lake Tekapo outlet flows were controlled and later diverted into a canal connecting with Lake Pukaki. In 1979, the Pukaki canal carrying water from Tekapo to Ohau A power station began operation (Robertson *et al.*, 1984).

The riverbeds are now used as spillways for flood flows. A small residual flow is maintained in the Tekapo riverbed. Its mean annual flow is roughly 10 cumecs (cf 80 before the canal was implemented). This combines downstream with groundwater from Forks stream, the Maryburn, Irishman's creek and other rainfed tributaries (pers. comm., E. J. Stead, National Institute of Water and Atmospheric

Research). The flow in the Pukaki River often now consists of groundwater only and seepage through the Pukaki canal and dam (pers. comm., Greg Carson, Waitaki Resource Consents Coordinator, ECNZ).

The Ohau River has been substantially modified by hydroelectricity works and only the lower 15 kilometres above Lake Benmore remains, the rest having been diverted into a canal or submerged by the artificial Lake Ruataniwha and Lake Benmore. Before completion of Ohau C dam in 1985, the Ohau river was carrying 80 cumecs of water from outflow through Ohau B station (ECNZ pamphlet). The residual flow in the lower Ohau is now less than 5 cumecs from seepage and it receives a further mean annual flow of about four cumecs from the rainfed Twizel tributary (Robertson *et al.*, 1984).

Study area.

The habitats studied are within the Tekapo and Ohau River deltas above Lake Benmore (Figure 2.1). Both deltas contain land which is willow covered, and areas which have been cleared of willows. The four areas chosen for study represent various conditions that have developed with different levels of disturbance from willow clearance and habitat reconstruction (see Figures 2.2 and 2.3). Several sites have been sampled within each area.

1 - the Ohau Control: Sites 1-3.

This habitat is within the Ohau River delta from 250 m - 900 m above Lake Benmore, below the confluence with the Twizel river. The Ohau riverbed is as close to a naturally braided state as is possible given the current level of management and this state is considered the goal of Project River Recovery. The river is occasionally flooded through release from Ruataniwha and due to unusual weather events. During the present study, it was flooded twice within a year (February 1993 and January 1994). Parts of the Ohau Control habitat have therefore been maintained at an early stage of plant succession. It was expected to show some changes in invertebrate and plant composition with time.

The lower Ohau was cleared by the Canterbury Regional Council in 1986 using chainsaws and a tractor. In 1987, the area was sprayed with Roundup and Pulse (10 l/ha) from a Hughes 300 helicopter, and spot sprayed for control of willow regrowth from the ground. It was spot sprayed again in 1990 (Roundup/Pulse) and subsequently in February 1994 with Grazon (10 l/ha) to kill broom (*Cytisus scoparius*) and willow regrowth (pers. comm., Bruce Scarlett, Canterbury Regional Council).

The vegetation on the riverbed ranges from sparse pioneer communities on shingle fans to patches of dense pasture on silty, pasture-like damp areas alongside the main channels.

Site 1 was a shingly, cobbled site with no vegetation on it. A thick (10 cm) layer of stones was deposited over all the trap holes during the January 1994 flood.

Site 2 was patchily covered by native pioneer riverbed vegetation such as *Epilobium melanocaulon*, *Muehlenbeckia axillaris*, *Elymus rectisetus*, mats of *Raoulia* spp. (up to 0.25 m across) and a few (introduced) broom plants (*Cytisus scoparius*) (Figure 2.4). Some by-kill of *Raoulia* spp. was observed after the latest spray application (pers. obs.) This site probably equates to the young (< 5 year old, grade 1-2) areas described in Burrows (1977).

Site 3 was a dense pasture-like silty area alongside the main channel. This moist site was dominated by adventive plants such as *Epilobium ciliatum*, *Mimulus guttatus* and *Juncus* spp., grasses and legumes (e.g. *Lotus pedunculatus*, *Trifolium repens*, *Festuca rubra*, and *Anthoxanthum odoratum*). This site was affected by both floods and was partly covered with stones (Figures 2.5 and 2.6).

The remaining study areas occurred in the Tekapo Riverbed from 250 m - 1600 m upstream from Lake Benmore.

2 - the Recovering Treatment area: Sites 10-12.

This habitat was within the Tekapo delta 250 m-800 m north of Lake Benmore.

At the onset of this study (December 1992), it was the most recently disturbed (cleared) area of riverbed. It was cleared in 1992 by the Canterbury Regional Council under contract to Department of Conservation. Willow trees were removed with diggers and bulldozed into windrows to dry. Windrows were sprayed with Grazon herbicide from a helicopter in October 1992 before reheaping in November of that year (pers. comm., Alicia Warren, Department of Conservation). Regrowth was sprayed with Roundup and Grazon in February 1993. In April, the heaps were burned and the remainder buried. The area was spot sprayed again with Grazon in November 1993.

Site 10 was a stony area sparsely covered by adventive shingle colonisers such as Californian poppy (*Eschscholzia californica*), hares-foot and white clover (*Trifolium arvense* and *T. repens*), pasture grasses (e.g. *Agrostis capillaris*) and regrowth from willow fragments (Figures 2.7 and 2.8).

Site 11 was a more silty site, sparsely covered by adventive pasture grasses, plantain (*Plantago lanceolata*), Californian poppy, and *Melilotus officinalis*.

Site 12 was more heavily vegetated and silty than site 11, and patchily covered with pasture grasses, plantain and legumes (especially white clover and *Lotus pedunculatus*).

All the sites were subjected to a high water table during a 165 cumec release of water down the Tekapo River for two weeks in January 1994.

3 - the main Experimental area: Sites 13-18.

This area was situated within the central Tekapo riverbed 800 m - 1600 m from Lake Benmore. Major changes in invertebrate and plant composition after bulldozing were expected within this area.

The area was cleared from 15 February to 19 March 1993. Trap sites were relocated as close to the original sites as possible using compass bearings taken from the pylons on the riverbed. Windrows were sprayed and reheaped in November.

At the commencement of this study, this area was similar in vegetation cover to the main Control zone (see below). Two types of habitat could be subjectively distinguished. These were:

1) the Shingly stony habitats (sites 13-15)

2) the grassy, Willow tree covered habitats (sites 15-18).

SHINGLE HABITAT:

Site 13 was a cobbled area with patches of silt and was bordered by distant willow trees. Vegetation was sparse comprising of a mixture of native pioneers and adventive opportunist plants such as sweet brier (*Rosa rubiginosa*), hawkweed (*Hieracium* spp), *Raoulia* mats, *Muehlenbeckia axillaris*, bryophytes and rock lichens. This site was affected by the flood in the Tekapo river; a thin layer of silt and green algae was deposited over it.

Site 14 was similar to site 13 in vegetation, but had a larger amount of cobblestones covered with rock lichens.

Site 15 was patchily covered by bryophytes and was one of the few sites where *Muehlenbeckia ephedroides* was recorded. It had fewer rocks covered by lichens, and more exposed hardened silt than site 14 (Figures 2.9 and 2.10).

WILLOW HABITAT:

Site 16 was a shaded site, covered with willow leaf-litter, pasture grasses and legumes, *Hieracium* sp., plantain, and remnants of *Muehlenbeckia axillaris*.

Site 17 was a shaded, damp site (one of the few places where *Coriaria* sp. was recorded) and received seepage from a river channel. *Muehlenbeckia axillaris* was in sward in this site and parasitised by dodder (*Cuscuta epithymum*). Other shaded

sward plants included *Lotus pedunculatus* and *Prunella vulgaris* (Figures 2.11 and 2.12). This site was partially covered by ponded water after January flood receded.

Site 18 was grassy with patches of silt, and covered by pasture grasses, sweet brier, plantain, white clover, and *Muehlenbeckia axillaris*.

4 - the main Control area: Sites 19-24.

This area represented an **uncleared** control (east of the Experimental habitat between the easternmost main Tekapo River channel and the Haldon Camp ground), 250 m - 1600 m from Lake Benmore. It was expected to show minimal changes in invertebrate and plant composition with time. Both Shingle and Willow habitat types were also distinguished in this area.

SHINGLE HABITAT:

Site 19 had a gravel/pebble substrate with patches of exposed silt. Vegetation included large (0.5 m diameter) *Raoulia* mats (with rabbit scratchings in the centre of some plants), *Racomitrium lanuginosum* and other bryophytes and some rock lichens.

Site 20 had more exposed silt than site 19, with patches of flannel-leaf (*Verbascum thapsus*), sweet brier and bryophytes.

Site 21 had patches of bare silt and pebbles partially covered by rock lichens, *Muehlenbeckia*, *Raoulia* and bryophytes (Figure 2.13).

WILLOW HABITAT:

Site 22 was a shaded site covered with pasture grasses such as *Agrostis capillaris* (browntop), *Festuca rubra*, *Poa* sp., and shrubs of sweet brier. This site was covered with water during the flood in January 1994; the remaining sites appeared to be unaffected.

Site 23 was covered primarily with sweet brier, browntop and sweet vernal (*Anthoxanthum odoratum*).

Site 24 was covered with sweet brier, pasture grasses, litter, legumes with an understorey of *Muehlenbeckia axillaris* (Figure 2.14).

Two additional areas within the Tekapo riverbed were also sampled. These represented riverbed flood channels cleared regularly since 1985 by the Ministry of Works and Canterbury Regional Council, and these areas were intermediate in condition between the Recovering Treatment and the Ohau habitat. Due to time constraints they were not included in the analysis of plants and invertebrates for this thesis.

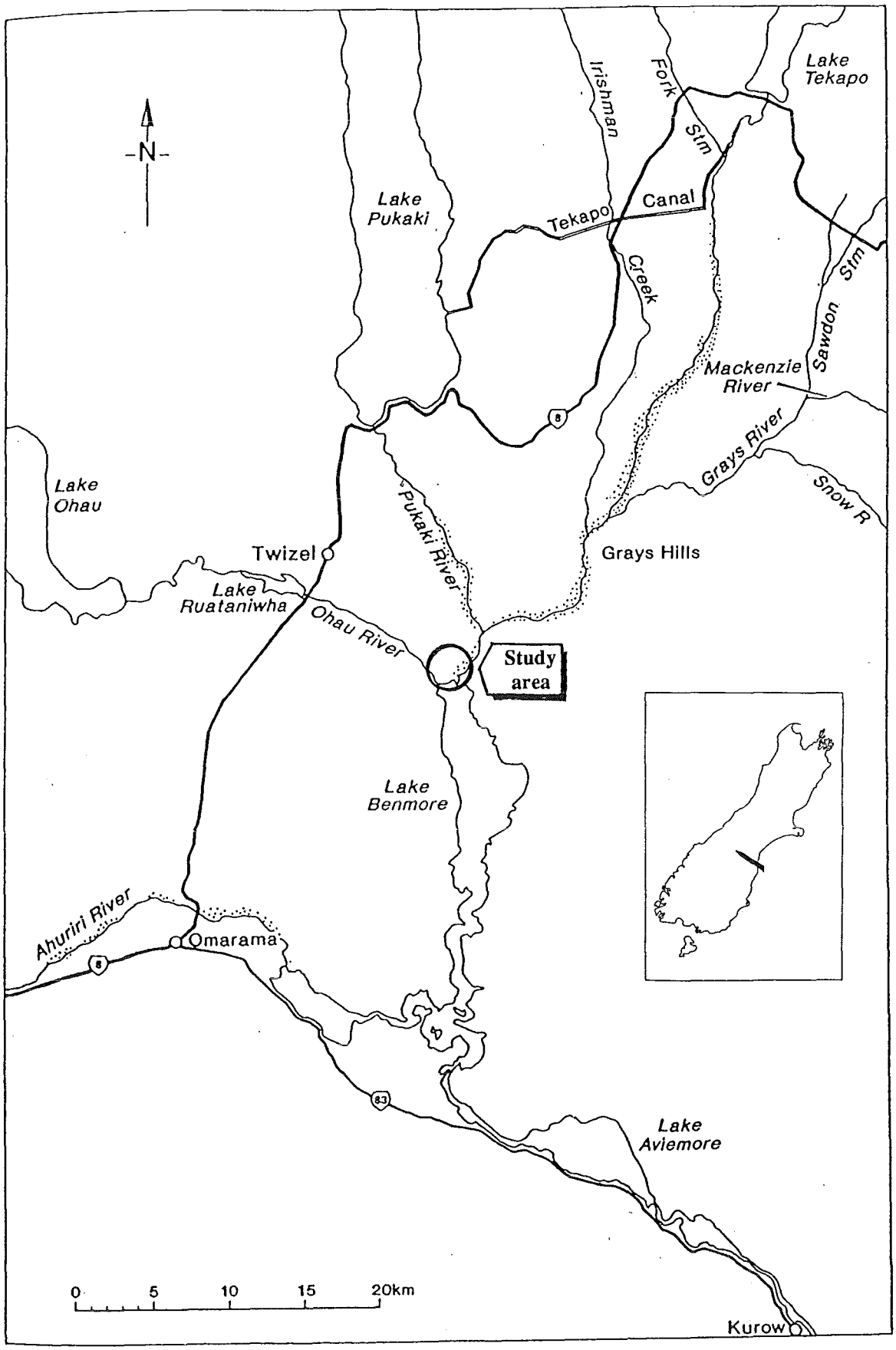

Figure 2.1. Map of the study area (from White, 1994).

Figure 2.2 and 2.3. Aerial photographs of the Tekapo (centre) and Ohau delta (left) in 1991 (pre-clearance) and 1993 (post-clearance). Green vegetation is predominantly crack willow (*Salix fragilis*) and sweet briar (*Rosa rubiginosa*). Brown rectangles in the second photograph are windrows of bulldozed trees. Numbers refer to site locations. (See Table 4.1.1a for site labels). Scale: 1cm = 100m

2.3

Figure 2.2 and 2.3. Aerial photographs of the Tekapo (centre) and Ohau delta (left) in 1991 (pre-clearance) and 1993 (post-clearance). Green vegetation is predominantly crack willow (*Salix fragilis*) and sweet briar (*Rosa rubiginosa*). Brown rectangles in the second photograph are windrows of bulldozed trees. Numbers refer to site locations. (See Table 4.1.1a for site labels). scale: 1cm = 100m

Figure 2.4. Site 2 from the Ohau habitat in 1993.

Figure 2.5. Site 3 from the Ohau habitat in 1993.

Figure 2.6. Site 3 from the Ohau habitat in 1994 (note flood damage).

Figure 2.7. Site 10 from the Recovering Treatment area in 1993 (bulldozed 1992).

Figure 2.8. Site 10 from the Recovering Treatment area in 1994.

Figure 2.9. Site 15 from the Experimental Shingle habitat before it was bulldozed (1993).

Figure 2.10. Site 15 from the Experimental Shingle habitat in 1994.

Figure 2.11. Site 17 from the Experimental Willow habitat before it was bulldozed (1993).

Figure 2.12. Site 17 from the Experimental Willow habitat in 1994.

Figure 2.13. Site 21 from the Control Shingle habitat in 1993.

Figure 2.14. Site 24 from the Control Willow habitat in 1993.

Chapter 3
Methods

Invertebrate sampling.

The study site was sampled on seven occasions:

6 - 9 December 1992;

* 16 - 19 January 1993;

8 - 11 April 1993;

17 - 20 September 1993;

30 November - 3 December 1993;

16 - 18 January 1994 (this sample was only partially completed before a flood warning was issued, so the collection was discontinued.)

* 11 - 14 February 1994 (taken immediately after flood waters had receded.)

For the purpose of this thesis, only the data from the two asterisked dates for the main untouched Control area, the Experimental (treatment) area, the Recovering Treatment area (cleared 1992) and the Ohau were used. This allowed a comparison of the treatment area before and after treatment with an untouched control area, a recently bulldozed area and a more braided riverbed habitat. All unused, unsorted invertebrate material and plant data sheets are held at Canterbury University by Mr. P. M. Johns (Zoology Dept).

Each of the areas was sampled using pitfall traps (see Appendix 2 for justification). Each trap was a plastic pottle (lip diameter of 110 mm) set flush with the ground. A strip of sand was laid around every trap hole to standardize trap accessibility for invertebrates. Traps were partially filled with a killing solution of 2 parts ethylene glycol to 1 part water (with a little detergent as a wetting agent). Trap holes in shingle areas were kept clear of debris with a tubular cardboard retainer of approximately the same dimensions as the pottle. Closed pitfall traps were placed in position for a day before they were set to minimise digging-in effects. Traps were opened and then left set for 3 consecutive days and nights.

Traps were arranged in the field in a hierarchical pattern. Five traps were positioned about 3 m from each other (sufficient distance for them to be relatively independent). Individual traps represented replicates within a **site**. Three Sites were treated as replicates within each **habitat**. The Experimental and Control areas each contained two habitats (one shingle and one willow) whereas the Ohau and Recovering Treatment area represented habitats in their own right. Each habitat contained 15 pitfall traps. In total, 90 traps were set during each sample period. Trap position was marked in the field with a numbered wooden stake (e.g. see Figure 2.4). Sites were noted on aerial photographs. After the Experimental area was bulldozed, trap sites were repositioned using compass bearings taken from high tension power pylons within the riverbed.

Basic weather observations are given in Appendix 1.

Sorting of invertebrates from pitfall traps.

After collection unsorted material from each pitfall trap was washed with water onto a 0.05 mm^2 sieve. The pore size of the sieve was sufficient to prevent all invertebrates visible under a 400 magnification from being washed away. The contents of the sieve were then flushed with 70 % ethanol via a funnel into a 3 x 1 cm glass vial and stored.

The contents of each tube (trap) were washed through a large pore (2 mm^2) sieve onto the smaller sieve, effectively separating "macro-organisms" from the "micro-organism and sand matrix". The "micro-matrix" contained sand and small organisms such as mites, Collembola and some Hymenopteran spider-egg parasitoids. The micro matrix was then washed back into the vial and stained with Eosine in 70% alcohol to enhance the contrast between cellular material and sand. Once stained, micro-organisms were decanted off the sand into a petri dish inscribed with a 1 cm grid. Counting of the "micro-matrix" was done by the author and two assistants.

Some taxa were subsampled (if there were more than about 80 specimens of that taxa per grid square) by counting only one quarter of the petri dish grid squares. A cardboard template with squares cut randomly from it (Figure 3.1) was used along

with substage and lateral lighting on a stereo microscope. This technique effectively illuminated the organisms inside the square and excluded (shaded) those outside the square. Specimens lying on the North and East boundaries were included in the count. This count multiplied by four represented the total count within that trap. Trials between assistants and between samples counted by each assistant showed the errors from subsampling and assistant discrepancies were less than ten percent.

Macro- and micro-organisms were separated into taxonomic groups, allocated an identification number, counted, then stored in 70% alcohol in 50 mm x 12 mm glass tubes. Most invertebrates were sorted to family level. Small and time-consuming organisms were grouped at their higher taxonomic classification levels. These were: all Thysanura; all Acari; larvae of Coleoptera, Lepidoptera and Diptera; the Dipteran families Sciaridae, Cecidomyiidae and Chironomidae (except for species 107); Trichoptera and Ephemeroptera. Three subjective divisions of Collembola were recognised *vis.* "Small grey" (Poduroidea), "Large stripy" (Entomobryoidea) and "Knobbly" (Neelipleona and Symphypleona). A reference collection of each taxon was developed to facilitate fast recognition and representative specimens were submitted to experts for identification. Mr. John Early (Auck. Inst. Museum) identified the Hymenoptera, Dr. Simon Pollard (Canterbury University) identified the Araneae, and Mr. Peter Johns identified the remainder.

Systematics primarily follow that in "Insects of Australia" (CSIRO, 1991) for the class, order and family designations. A list of taxa is given in Appendix 4.

Survey of the trap environment.

A comprehensive vegetation survey was conducted in January 1993 and repeated in January/February 1994 (post clearance) to investigate and quantify the immediate environment around each pitfall trap. A preliminary list of plant taxa was compiled from specimens taken from Tekapo and Ohau riverbeds in December 1992. Each plant was allocated an identification number. Voucher specimens of plants with uncertain identification were collected. All vascular plants were identified by Mr M. F. Sinclair. A list of plants is given in Appendix 5.

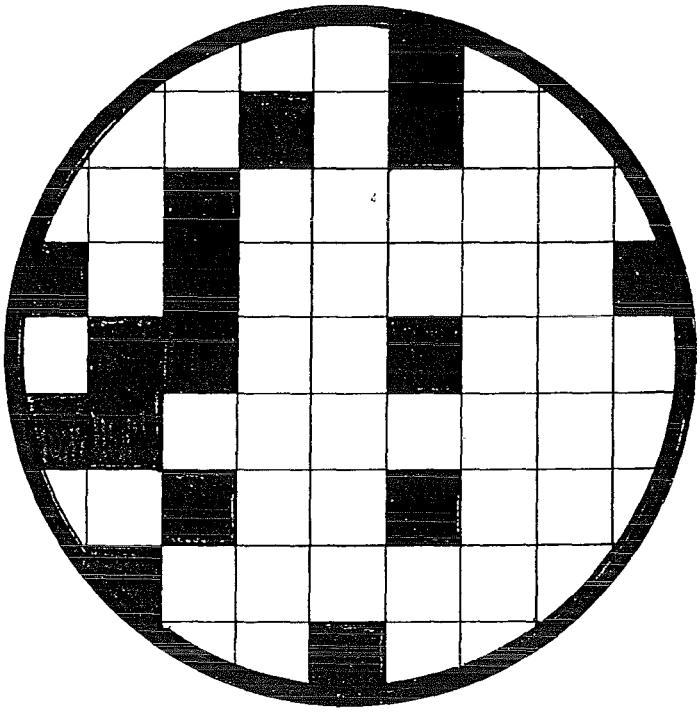

Figure 3.1. Diagram of the petri-dish template used in subsampling the 'micro-matrix'. Black squares denote holes in the template (actual size).

Four quadrats (each 1 m x 1 m) were oriented to the cardinal points in a crucifix around every pitfall trap hole. The central square containing the vacant trap hole was not sampled. Within each quadrat, the percentage cover of each plant species was estimated to the nearest five percent. The category of < 5 % was the lowest percentage cover score possible for plants that were represented. Scores were adjusted slightly *a posteriori* so that the taxa in the < 5 % category contributed a minimal amount to the total plant cover. Bryophytes and lichens were categorized into broad guilds, (e.g. sand lichen, brown moss) with a few exceptions (e.g. *Racomitrium* sp, Teal rock lichen).

Environmental parameters were also recorded within each quadrat. These were:

1) - percentage cover of bare ground, which was divided into six particle size categories:

Silt < 0.1 mm; Sand 0.2 mm - 2 mm; Gravel 3 mm - 16 mm; Pebbles 17 mm - 6.3 cm; Cobbles 6.4 cm - 25 cm; Boulders > 25 cm.

2) - litter percentage cover and depth (cm);

3) - lagomorph (rabbit)-dropping percentage cover;

4) - proximity to willow trees or other major features (e.g. river);

5) - shade (0, quarter, half, three quarter, full);

6) - miscellaneous effects (e.g. extent of silt cover after flooding).

'Total Percentage of ground cover' was recorded to summarise the data and included the total percentage cover for vascular plants, lichens, bryophytes, rabbit dropping and litter. The Bare ground score was the ground cover score subtracted from 100 %.

Three field assistants and the author worked in pairs to quantify the trap environment. M. F. Sinclair and the author identified plant species in the field, and estimated percentage cover of all the parameters while the remaining two assistants recorded the information, estimated willow proximity and drew major features of the plot on the record sheet. To maintain consistency, the same people recorded the same

plots when the survey was repeated.

Analyses of Data.

Plant and invertebrate data from each trap were entered into the spreadsheet Quattro Pro version 5.0, and the total per site (i.e. sum of 5 traps) was calculated. The plant and environmental data was summed across each site (20 quadrats) then averaged.

Data were investigated from two perspectives:

1) Site, habitat, and community data were analyzed using simple and complex statistical methods to investigate the sensitivity of each method for distinguishing between habitat types and detecting a treatment effect;

2) Plant and invertebrate data were investigated at a species level to determine the following; whether there was a significant improvement in habitat quality of the bulldozed areas (change towards a braided state); whether absence of taxa in the 1994 sample could be attributed to bulldozing; whether invertebrate prey of mammalian predators had changed in availability; whether invertebrates of conservation value existed and the impact of bulldozing on them.

SIMPLE METHODS:

Species richness, evenness/equitability and diversity.

Species richness was calculated for invertebrates and plants on a site and habitat level. Chi-square tests of significance were performed for each type of data to determine the difference in species richness between sample dates. Richness scores per habitat were also compared within each sample date. Species evenness and diversity were only calculated for invertebrate data. Equitability and diversity per site and habitat were calculated using the reciprocal of the Two Dominant Species Index and Margalef's index respectively.

Relative distribution of invertebrate taxa across five orders (per habitat)

The relative number of species within five major invertebrate taxa per habitat was calculated. Taxa were grouped into Coleoptera, Diptera, Hymenoptera,

Hemiptera, Araneae and a miscellaneous category which included all other invertebrate orders present (e.g. Isopoda, Lepidoptera).

Temporal stability (persistence) between sample dates of plants and invertebrates.

This was calculated by subtracting the number of taxa lost and gained (from site or habitat) across the two dates from the overall number of taxa found (total for both dates combined). The number remaining represented the number of taxa shared between sample dates (an indication of persistence).

COMPLEX METHODS:

Classification and Ordination.

Community classification and ordination of invertebrate and plant data was carried out in PATN. The total abundance of invertebrates per site were calculated (sum of five traps) and data were $\log_{10}$ transformed. The mean of plant percentage cover and environmental parameters per site was calculated (average across 20 quadrats) and data was $\log_e$ transformed. This gave a comparable range of scores between the vegetation and invertebrate data.

Sites were classified using an agglomerative clustering technique. The Bray-Curtis measure of dissimilarity was used and sites were fused with UPGMA (average linkage Beta = 0). Sites were grouped to a level where the main habitat types ('Willow', 'Shingle', 'Ohau' and 'bulldozed/disturbed') were still roughly apparent. Species and sites were arranged in a two-way table (with global standardisation which represents species abundance on a scale of 1 to 9. Species were arranged across the table by the programme, which used the Kruskal Wallis test of significance.

Ordination of plant and invertebrate data was carried out using CANOCO. Invertebrate data and plant data were ordinated separately with DCA (using the default settings) (see Appendix 3). Invertebrate data were ordinated again (using DCCA) with inclusion of plant species data as a secondary matrix. The invertebrate ordination was then repeated with the 'macro' plant percentage cover scores as well as the environmental data included as a secondary matrix. Comparison of the latter

two ordinations was made to determine which level of survey would be a sufficient (better) predictor of invertebrate species composition.

The effect of bulldozing on invertebrate and plant taxa was also investigated at a species level.

Improvement of habitat quality after bulldozing.

Invertebrate and plant taxa were investigated to determine if the bulldozed habitats changed towards a braided riverbed state. This was measured by the appearance of 'braided riverbed' taxa in the Experimental and Recovering Treatment habitats before and after bulldozing.

Species which occurred in the Ohau habitat were considered 'braided riverbed' taxa. A list was made of those 'braided riverbed' taxa present in the Experimental habitats before bulldozing. Taxa which also occurred in other habitats were subtracted from the list. The remaining species represented 'braided riverbed' taxa already present in the Experimental habitats before willow clearance.

A list was made of 'braided riverbed' taxa occurring in the bulldozed habitats. The 'braided riverbed' taxa which occurred in other (non-treatment) habitats, plus those taxa which appeared in the Control habitats in 1994 only (seasonal species), plus the taxa occurring before as well as after bulldozing (resilient taxa) were subtracted from the list. The remaining species represented 'braided riverbed' taxa which appeared in the Experimental habitats after clearance. Species which were represented in the Ohau habitat from site 3 only were noted as these could represent riparian invertebrates available as prey to Wrybills.

Loss of plant and invertebrate taxa from the Experimental habitats.

The absence of taxa in the 1994 sample of the bulldozed habitats was examined to determine if their absence could be a result of willow clearance. Taxa were listed and their loss, gain or continued presence in other habitats was noted. Those species present in Control habitats in 1993 but absent in 1994 were considered

seasonal losses. Taxa which were present in 1994 but not 1993 were considered seasonal gains. Taxa which were lost from the Experimental habitats but remained present in other habitats were considered to have been negatively affected by bulldozing.

Availability of invertebrate prey for mammalian predators of riverine birds.

Invertebrates consumed by predators were identified by the author from cat and ferret gut and faecal material supplied by Amelia Pascoe (MSc student, University of Otago). Invertebrate fragments were identified and the number of specimens were counted where possible. The impact of bulldozing on invertebrate prey species was assessed by comparing their abundance in the 1993 sample with their abundance in the 1994 sample.

Invertebrates of conservation value.

The invertebrate taxa from the two analyzed samples and the remaining unsorted samples were scrutinized to detect species of conservation value. Macro-taxa which were rare, endemic, restricted in distribution or undescribed were considered important, and the impact of bulldozing on them was assessed.

Chapter 4
Results

In total, 233 invertebrate taxa and 117 plant taxa were recorded from the analyzed samples. Lists of taxa are in Appendix 4 and 5. Site totals for each taxa are appended in a 720k disk (Appendix 6). Plant and environmental data are total percentage cover scores per site (from 20 quadrats) and invertebrate data are totals from each site (five pitfall traps). The data was saved in Quattro pro (version 5) as a text file and can be accessed through an IBM compatible programme.

ANALYSIS OF COMMUNITY DATA WITH SIMPLE STATISTICAL METHODS

4.1. Species Richness

Per site

Site 1 (Ohau) had the lowest number of invertebrate taxa, while site 18 (Experimental Willow post-treatment) recorded the most (Figure 4.1.1). Overall, an increase in species richness from the 1993 sample to 1994 sample was recorded from 12 out of all 18 sites, but only the increase in site 13 (Experimental Shingle) and site 10 (Recovering Treatment area) was significant (Table 4.1.1; site and habitat codes are shown in Table 4.1.1a).

Site 1 in the Ohau delta had no plant species on it, and site 3 (the 1993 sample) had the most (Figure 4.1.2). 11 out of the 18 sites increased in plant richness from 1993 to 1994, but only the increase in site 18 and the decrease in site 3 were significant.

When plant and invertebrate species richness were grouped together (total richness per site), the Ohau site 1 still had the lowest species richness while site 3 (in 1993) the highest. The Ohau site 3 showed a significant decrease in taxa (as a result of flood damage) and a significant increase in taxa was recorded in three Experimental sites (13, 14 and 18) (Figure 4.1.3).

Per habitat

An increase in invertebrate taxa was recorded for all habitats (Figure 4.1.4), but only the increase in the Experimental Shingle habitat was significant (Table 4.1.2). In 1993, the Experimental Willow habitat had the highest richness followed by the Control Willow habitat, the Ohau, and Recovering Treatment. Both the Experimental and Control Shingle habitats had a lower species richness.

In 1993 the species richness recorded for the Experimental Willow habitat was significantly higher than all habitats except the Ohau (Table 4.1.3). The Control Willow was only significantly higher than the Experimental Shingle habitat.

Both the Control and the Ohau habitats changed little in their invertebrate species richness from 1993 to 1994 (Table 4.1.4); the increase in the number of species in the Recovering Treatment habitat meant that it became significantly different from the Control Shingle. The Experimental Shingle habitat also gained species to the extent that it was no longer significantly different in the number of taxa from the Willow habitats and the number of taxa was significantly higher than that of the Control Shingle habitat. The small increase in the number of taxa found in the Experimental Willow habitat meant that it became significantly different in richness from the Control Shingle and Ohau habitats.

Except for the Ohau habitat, an increase in the number of plant taxa after treatment was recorded. The Ohau habitat had a significant decrease in taxa as a result of flooding (Table 4.1.2 and Figure 4.1.5). The Ohau had the greatest number of plant taxa in 1993, followed by the Experimental Willow, Recovering Treatment habitat then the Experimental Shingle and Control Shingle habitats (Table 4.1.5). Both the Control habitats (Shingle and Willow) had a low species richness. These habitats were significantly lower than the Experimental Willow, Control Willow and Recovering Treatment habitats on both sample dates, and significantly lower than the Ohau before it was flooded (1993). In contrast to the invertebrate richness, the Experimental Shingle habitat did not show a significantly lower plant species richness from the Experimental Willow habitat, and this difference decreased after bulldozing (Table 4.1.6).

When the total number of taxa are considered (plants plus invertebrates) at the habitat level, only the Experimental Shingle habitat had a significantly higher richness in 1994 (Table 4.1.2), although all habitats (except the Ohau) increased in the number of taxa (Figure 4.1.6).

*Table 4.1.1. Species richness scores (S) and Chi-square significance tests for sites: Inv = Invertebrate; Plt = Plant. Blank cells denote not significant at 0.05 level, () denote close to significant, * and ** = significant at 0.05 and 0.01 level respectively. Site and habitat codes are shown in Table 4.1.1a.*

Site	Habitat	Inv S '93	Inv S '94	Chi-square	Plt S '93	Plt S '94	Chi-square	Total S '93	Total S '94	Chi-square
1	Ohau	17	23		0	0		17	23	
2		29	25		15	17		44	42	
3		50	54		62	34	**	112	88	*(*)
10	Rec Trt	27	46	*	40	40		67	86	
11		30	35		34	34		64	69	
12		44	53		44	46		88	99	
13	Expt Sh	28	53	**	34	37		62	90	*
14		31	42		26	39		57	81	*
15		34	32		29	32		63	64	
19	Cont Sh	31	37		26	26		57	63	
20		30	34		25	28		55	62	
21		32	27		25	30		57	67	
16	Expt W	54	53		34	36		88	89	
17		49	51		37	34		86	85	
18		47	56		28	47	*	75	103	*
22	Cont W	54	45		25	27		79	72	
23		39	45		21	24		60	66	
24		48	48		31	30		79	78	

Table 4.1.1a. Site and habitat codes. (Colours denote habitats in site ordinations).

1993 Site code	1994 Site code	Habitat abbreviation	Habitat
2-1	7-1	Ohau	Ohau control
2-2	7-2		Ohau control
2-3	7-3		Ohau control
2-10	7-10	Rec Trt	Recovering Treatment
2-11	7-11		Recovering Treatment
2-12	7-12		Recovering Treatment
2-13	7-13	Expt Sh	Experimental Shingle
2-14	7-14		Experimental Shingle
2-15	7-15		Experimental Shingle
2-16	7-16	Expt W	Experimental Willow
2-17	7-17		Experimental Willow
2-18	7-18		Experimental Willow
2-19	7-19	Cont Sh	Control Shingle
2-20	7-20		Control Shingle
2-21	7-21		Control Shingle
2-22	7-22	Cont W	Control Willow
2-23	7-23		Control Willow
2-24	7-24		Control Willow

Table 4.1.2. Invertebrate, plant, and total richness scores per habitat. Chi-square levels of significance between samples (1993 vs 1994). Blank cells denote not significant at the 0.05 level; () = close to significant; * and ** = significant to the 0.05 and 0.01 level respectively. (See Table 4.1.1a for habitat labels).

	Invertebrates	Plants	Total 1993 S	Total 1994 S	Total significance
Ohau		*	132	112	
Rec Trt			121	144	
Expt Sh	**		101	141	*
Cont Sh			93	100	
Expt W			146	154	
Cont W			115	118	

Table 4.1.3. Invertebrate species richness scores per habitat in 1993; Chi-square levels of significance amongst richness scores. Bracketed numbers in column labels are species richness scores per habitat. Blank cells = not significant at 0.05 level, () = close to significant at 0.05 level, * and ** = significant at 0.05 and 0.01 level respectively.

	Ohau (66)	Rec Trt (63)	Expt Sh (53)	Cont Sh (58)	Expt W (87)	Cont W (78)
Rec Trt						
Expt Sh						
Cont Sh						
Expt W		*	**	*		
Cont W			*			

Table 4.1.4. Invertebrate species richness scores per habitat in 1994; Chi-square levels of significance amongst richness scores. Bracketed numbers in column labels are species richness scores per habitat. Blank cells = not significant at 0.05 level, () = close to significant at 0.05 level, * and ** = significant at 0.05 and 0.01 level respectively.

	Ohau (67)	Rec Trt (84)	Expt Sh (84)	Cont Sh (59)	Expt W (93)	Cont W (79)
Rec Trt						
Expt Sh						
Cont Sh		*	*			
Expt W	*			**		
Cont W						

Table 4.1.5. Plant species richness scores per habitat in 1993; Chi-square levels of significance amongst richness scores. Bracketed numbers in column labels are species richness scores per habitat. Blank cells = not significant at 0.05 level, () = close to significant at 0.05 level, * and ** = significant at 0.05 and 0.01 level respectively.

	Ohau (66)	Rec Trt (58)	Expt Sh (48)	Cont Sh (35)	Expt W (59)	Cont W (37)
Rec Trt						
Expt Sh						
Cont Sh	**	*				
Expt W				*		
Cont W	**	*			*	

Table 4.1.6. Plant species richness scores per habitat in 1994; Chi-square levels of significance amongst richness scores. Bracketed numbers in column labels are species richness scores per habitat. Blank cells = not significant at 0.05 level, () = close to significant at 0.05 level, * and ** = significant at 0.05 and 0.01 level respectively.

	Ohau (45)	Rec Trt (61)	Expt Sh (57)	Cont Sh (40)	Expt W (61)	Cont W (39)
Rec Trt						
Expt Sh						
Cont Sh		*				
Expt W				*		
Cont W		*			*	

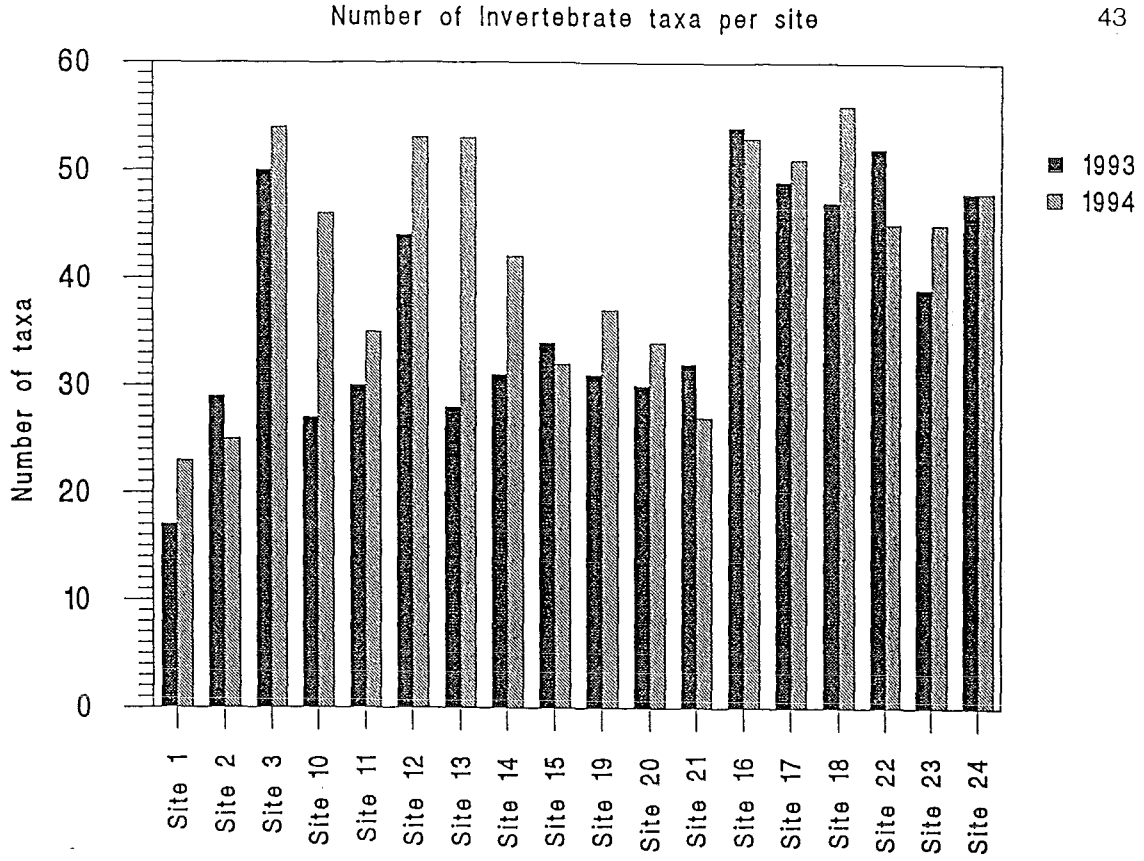

Figure 4.1.1. Number of invertebrate taxa per site (total from five pitfall traps) for two sample dates. Experimental sites were bulldozed after the first sample. Sites 1-3 are from the Ohau habitat (flooded after first sample); 10-12 from Recovering Treatment area (bulldozed 1992); 13-15 from Experimental Shingle habitat; 16-18 from Experimental Willow; 19-21 from Control Shingle (undisturbed); 22-24 from Control Willow (undisturbed).

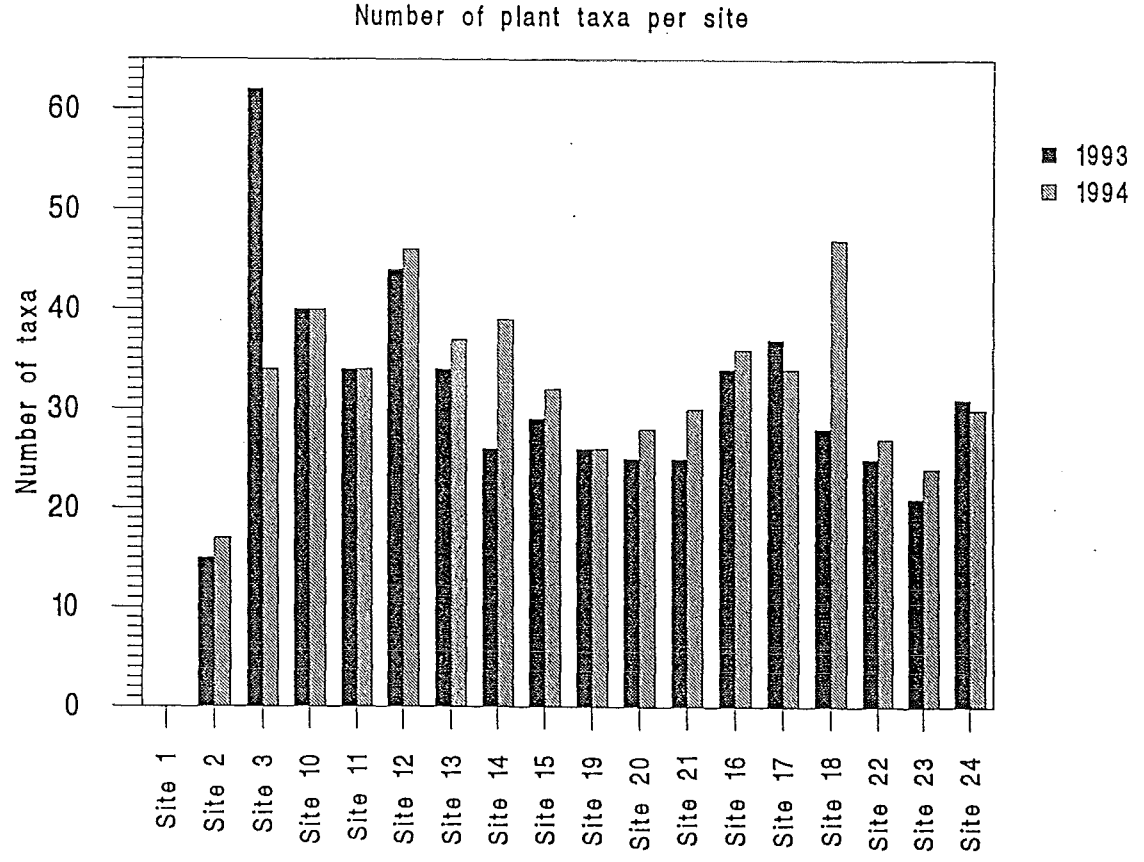

Figure 4.1.2. Number of plant taxa per site (total from 20 quadrats) for two sample dates. Experimental sites were bulldozed after the first sample. Sites 1-3 are from the Ohau habitat (flooded after first sample); 10-12 from Recovering Treatment area (bulldozed 1992); 13-15 from Experimental Shingle habitat; 16-18 from Experimental Willow; 19-21 from Control Shingle (undisturbed); 22-24 from Control Willow (undisturbed).

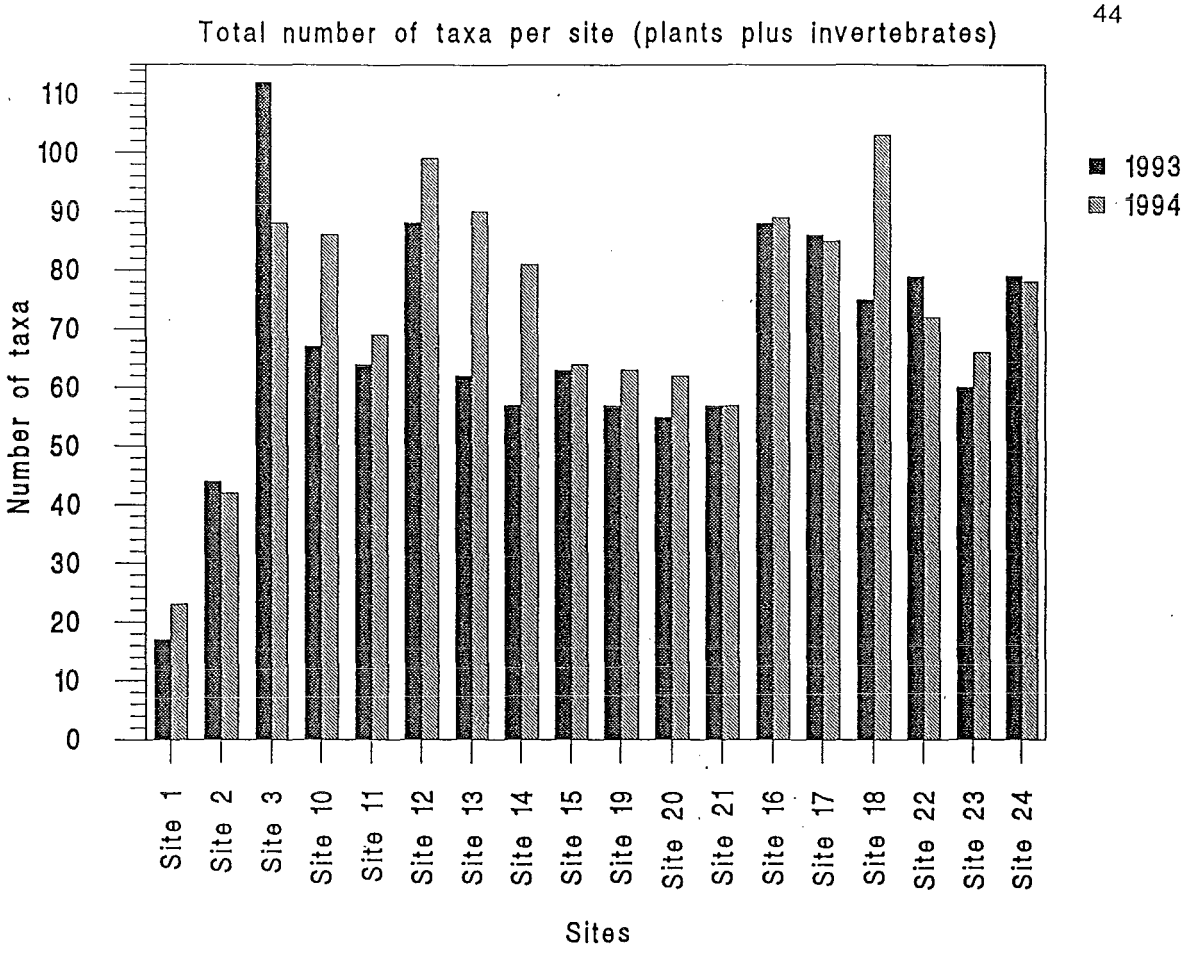

Figure 4.1.3. Total number of taxa per site (plant plus invertebrates). Experimental sites were bulldozed after the first sample. Sites 1-3 are from the Ohau habitat (flooded after first sample); 10-12 from Recovering Treatment area (bulldozed 1992); 13-15 from Experimental Shingle habitat; 16-18 from Experimental Willow; 19-21 from Control Shingle (undisturbed); 22-24 from Control Willow (undisturbed).

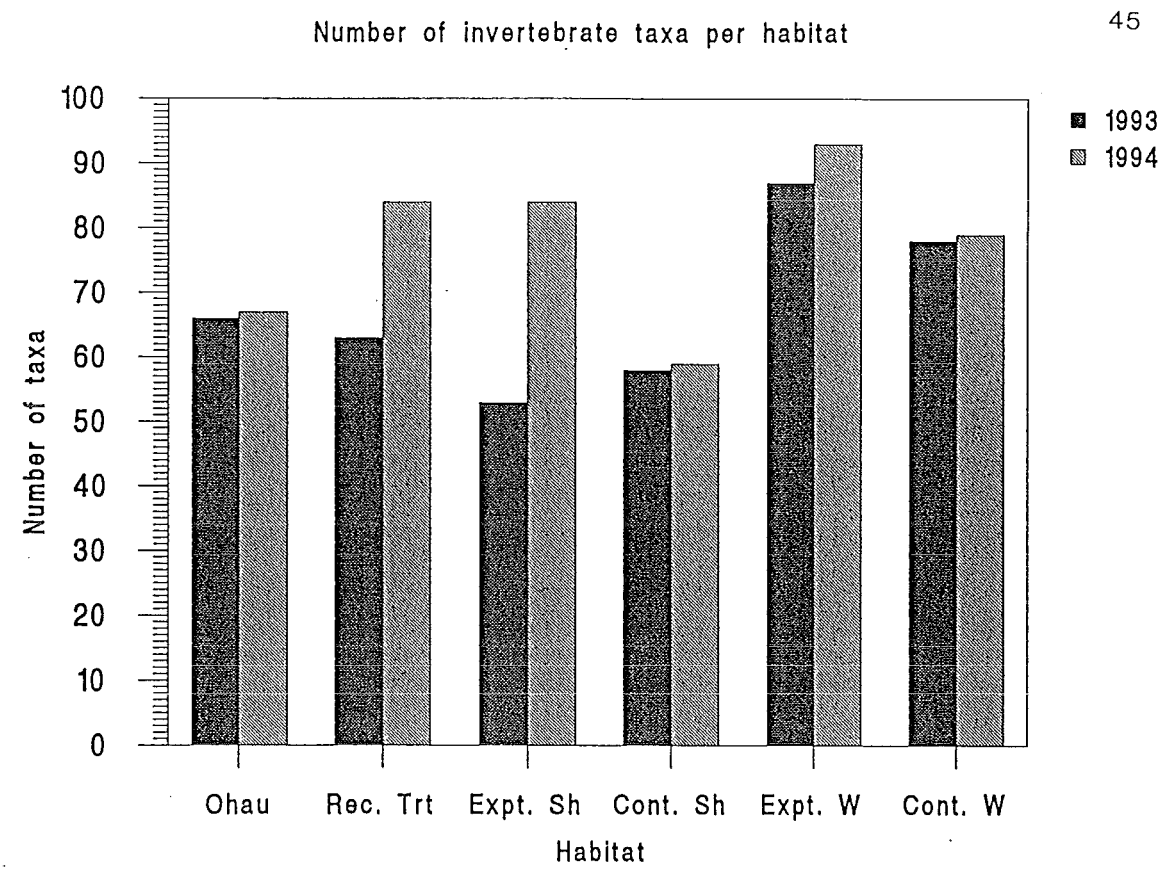

Figure 4.1.4. Number of invertebrate taxa per habitat (total across three sites). Experimental sites were bulldozed after the first sample. (See Table 4.1.1a for habitat labels).

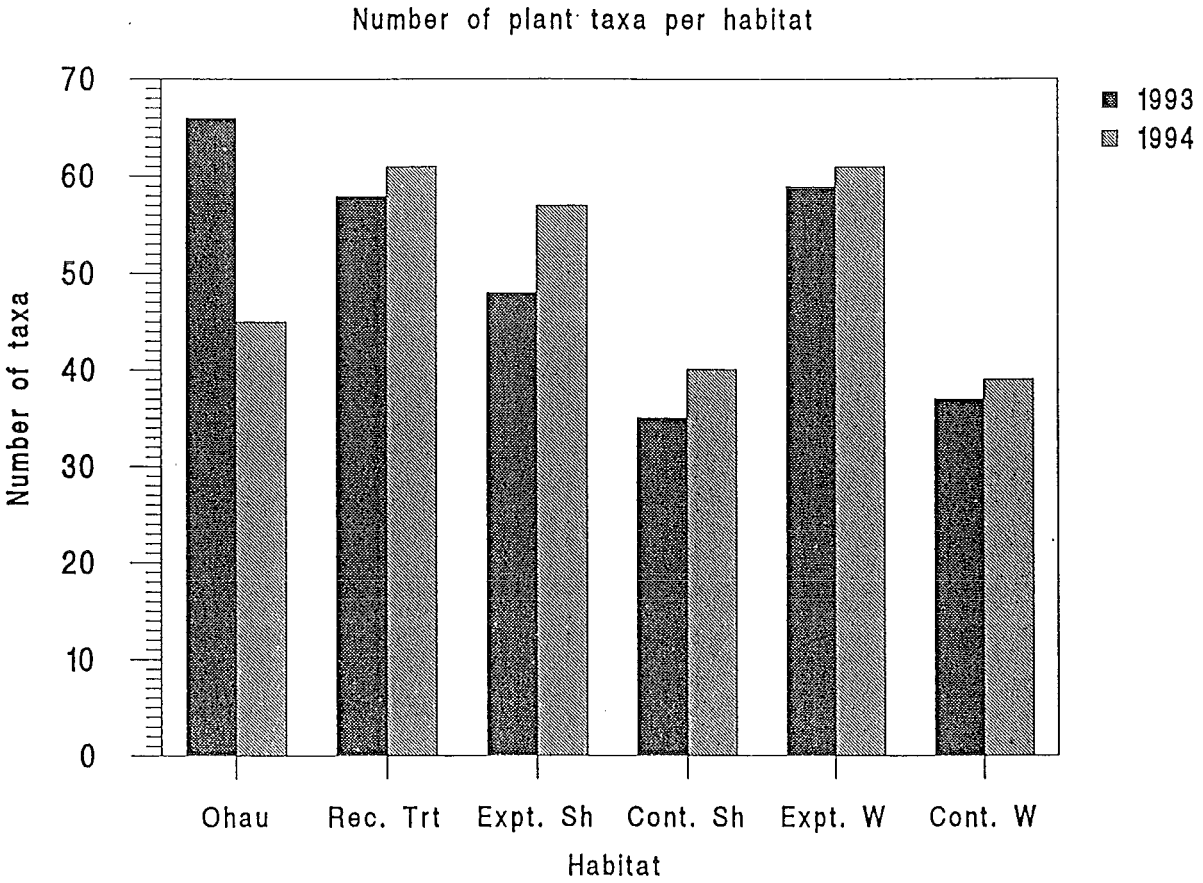

Figure 4.1.5. Number of plant taxa per habitat (total across three sites). Experimental sites were bulldozed after the first sample. (See Table 4.1.1a for habitat labels).

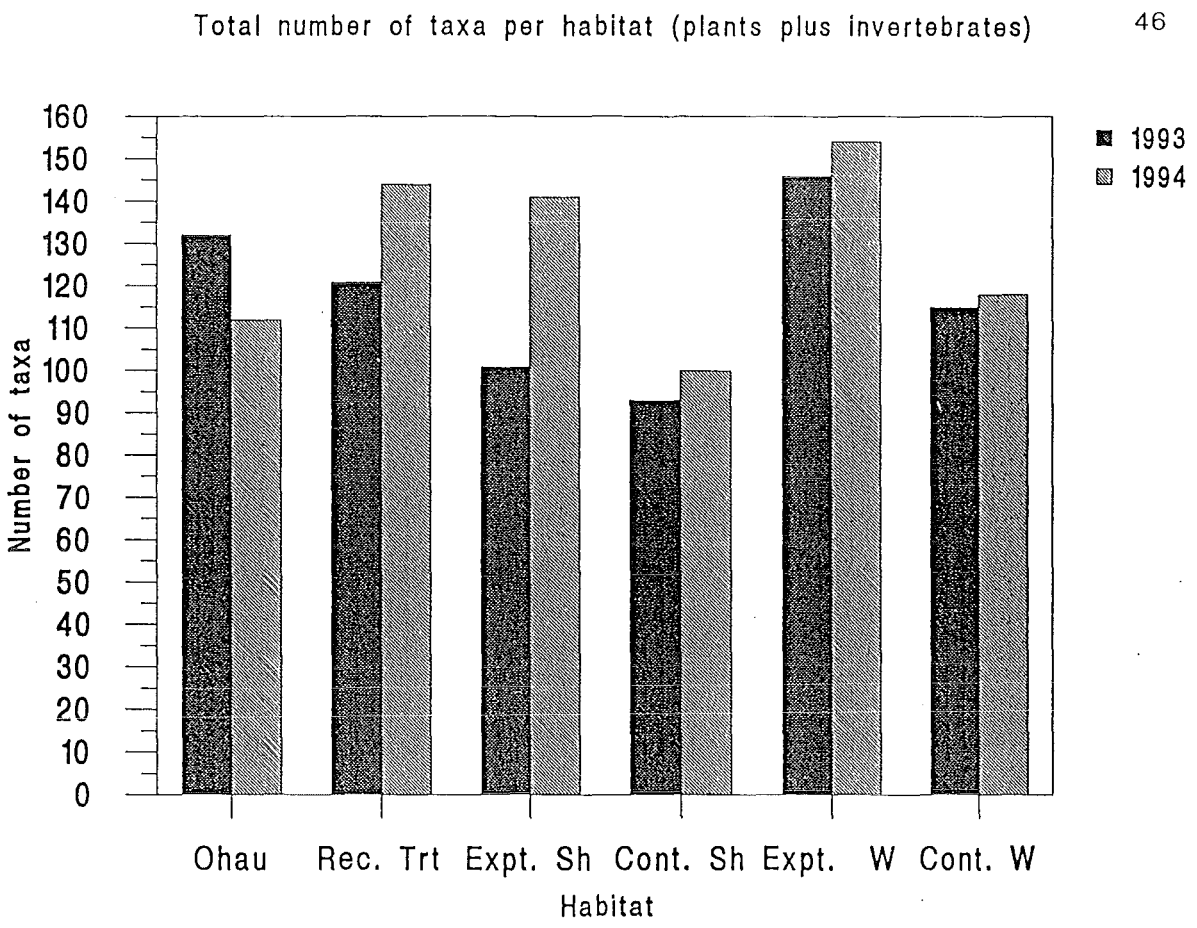

Figure 4.1.6. Total number of taxa per habitat (plants plus invertebrates). Experimental sites were bulldozed after the first sample. (See Table 4.1.1a for habitat labels).

4.2. Invertebrate equitability (evenness)

Per site

The dominant species in the majority of sites were Collembola and mites (Table 4.2.1 and Figure 4.2.1). The range of evenness scores per site was low. The highest score was recorded from Site 22 (Control Willow in 1994) while sites 13 (Recovering Treatment area), 20 and 21 (Control Shingle) had the lowest scores and therefore the spread of individuals across the species was less even.

Per habitat

When grouped at a habitat level the Ohau and Recovering treatment habitats in 1993 had slightly higher values than the other habitats on that date (Table 4.2.2 and Figure 4.2.2). All the 1994 scores were higher than the 1993 scores. The Recovering Treatment area had the highest score in the 1994 sample. After bulldozing, the Experimental Shingle and Willow habitats had a slightly higher scores which indicated a more even distribution of individuals across the species than in the 1993 samples. The Control Shingle was consistently low between years, although the 1994 score for this habitat was slightly elevated.

4.3. Species diversity

Per site

Site 22 (Willow Control) had the highest diversity score in 1993 and 1994 (Table 4.3.1). It had the same species richness as the 1993 sample of site 16, but fewer specimens which reduced its overall diversity below that of site 16. Sites 3 of the Ohau and site 12 of the Recovering Treatment area had high diversities (Figure 4.3.1) and site 13 (Recovering Treatment) and site 20 (Control Shingle) had the lowest diversity scores. Site 20 and site 1 had similar diversity scores despite the huge specimen count and almost double the number of species in site 20 than in site 1.

Per habitat

When grouped at the habitat level, the diversity scores more or less reflected the species richness scores (see section 4.1). All the 1994 diversities per habitat were higher (even in the control zones) than the 1993 sample (Figure 4.3.2). Overall, the Control Willow habitat had the highest diversity (in the 1994 sample) while the Experimental Shingle (before bulldozing) had the lowest (Table 4.3.2). A decrease in abundance (primarily of 'small grey' Collembola from 19462 specimens to 222 specimens) meant that the 1994 sample of the Control Willow habitat had a higher diversity than the 1993 sample from that same habitat, despite them both having similar species richness scores.

In general, the Shingle habitats were lower in diversity than the other habitats in 1993 (through a combination of a high number of specimens and a low species richness). The Experimental shingle habitat increased in diversity after bulldozing. The Control Willow sample in 1993 was lower in diversity than the Experimental Willow habitat in 1993, but in 1994 the differences were minimal.

Table 4.2.1. Invertebrate species Evenness/equitability per site (Inverse of Two Dominant Species index) The 'Dominant spp' refers to the following taxa which were dominant in each site; 1 = 'small grey' Collembola; 2 = mites; 3 = 'large stripy' Collembola; 4 = *Actenonyx bembidioides* (Carabidae); 5 = Dictynidae sp 2 (cursorial spider); 6 = 'knobbly' Collembola; 7 = *Ostenia*? sp (Diptera); 8 = *Phalangium opilio* (harvestman).

Site	1993		1994	
	Dominant spp	Score	Dominant spp	Score
1 Ohau	4, 8	1.5625	2, 3	1.1764
2	1, 2	1.2345	2, 3	1.2195
3	1, 5	1.4705	2, 6	1.3157
10 Rec Trt	2, 3	1.2345	6, 2	1.4084
11	2, 3	1.1627	2, 3	1.1235
12	2, 3	1.5384	6, 1	1.7857
13 Expt Sh	1, 2	1.0101	1, 2	1.2820
14	1, 2	1.0638	2, 3	1.2195
15	1, 3	1.3513	3, 2	1.0526
19 Cont Sh	1, 2	1.0869	3, 2	1.0752
20	1, 3	1.0101	3, 2	1.0752
21	1, 2	1.0101	2, 3	1.1235
16 Expt W	1, 2	1.0416	6, 2	1.5625
17	1, 2	1.3888	2, 6	1.1764
18	1, 2	1.0989	6, 2	1.4705
22 Cont W	1, 2	1.3157	2, 7	2
23	1, 2	1.0204	2, 3	1.5151
24	1, 2	1.0204	2, 7	1.4492

Table 4.2.2. Invertebrate species evenness/equitability per habitat, measured by the inverse of the Two Dominant Species Index. (See Table 4.2.1 for Dominant species codes.

Habitat	1993		1994	
	Dominant spp	Score	Dominant spp	Score
Ohau	1, 2	1.2345	2, 3	1.3698
Rec Trt	2, 3	1.2658	2, 6	1.6666
Expt Sh	1, 2	1.0204	2, 3	1.6129
Cont Sh	1, 2	1.0204	3, 2	1.0869
Expt W	1, 2	1.0989	2, 6	1.3888
Cont W	1, 2	1.0309	2, 3	1.5873

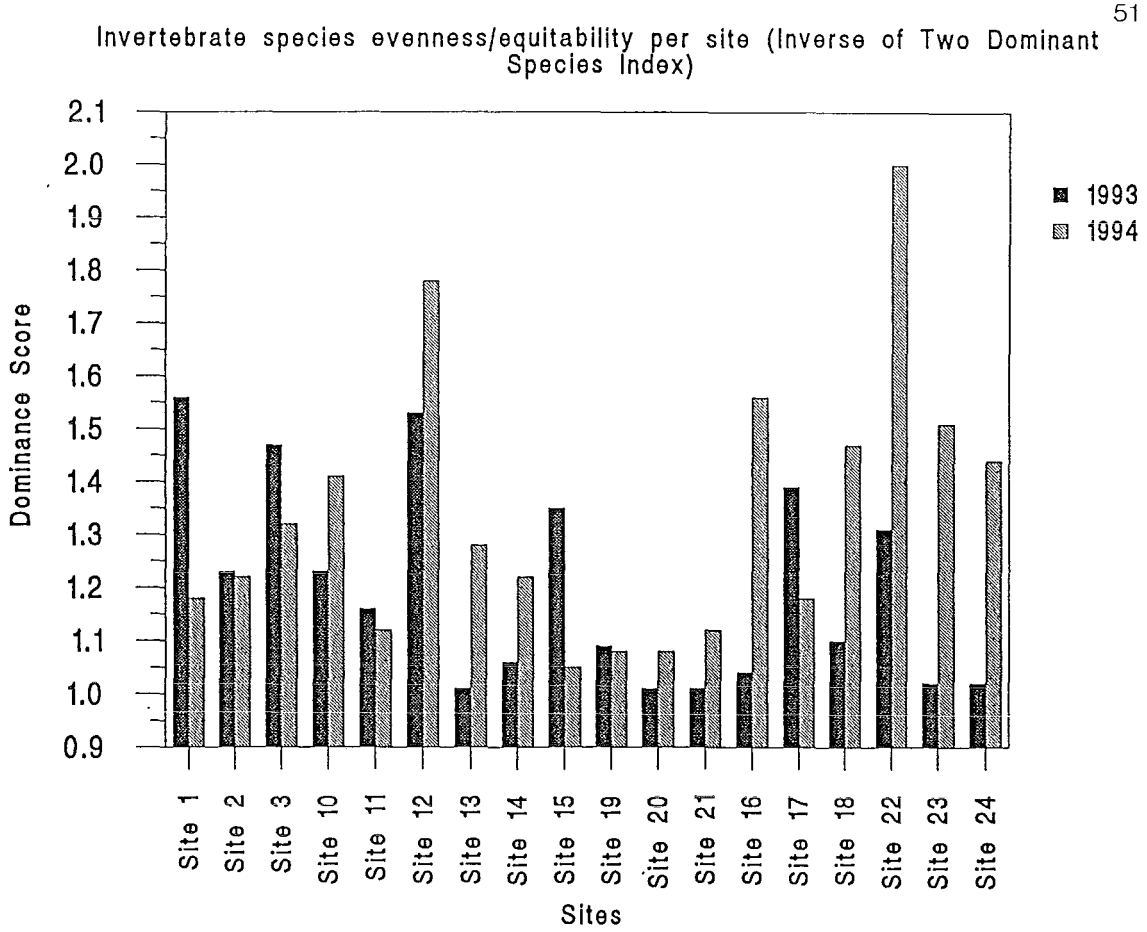

Figure 4.2.1. Species evenness/equitability of invertebrate taxa per site, measured by the inverse of the Two Dominant Species Index. Experimental sites were bulldozed after the first sample. Sites 1-3 are from the Ohau habitat (flooded after first sample); 10-12 from Recovering Treatment area (bulldozed 1992); 13-15 from Experimental Shingle habitat; 16-18 from Experimental Willow; 19-21 from Control Shingle (undisturbed); 22-24 from Control Willow (undisturbed).

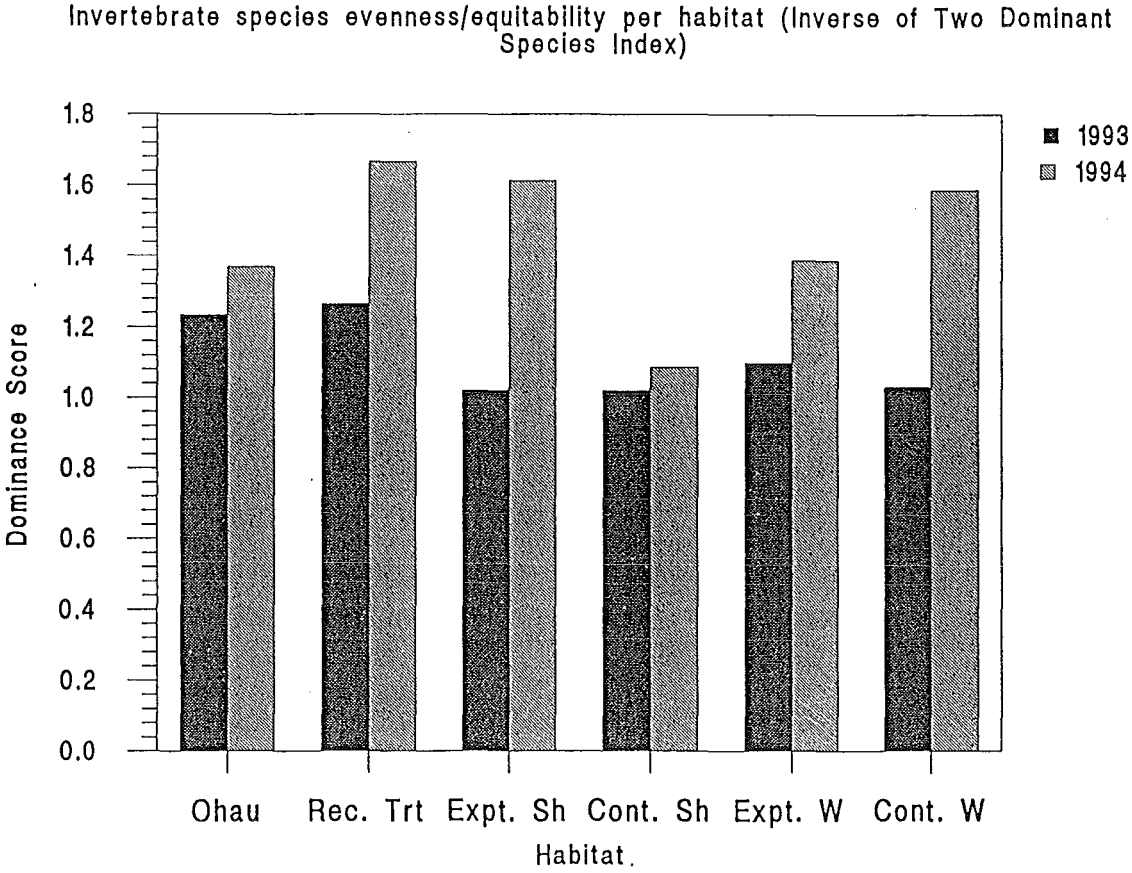

Figure 4.2.2. Species evenness/equitability of invertebrate taxa per habitat, measured by the inverse of the Two Dominant Species Index. (See Table 4.1.1a for habitat labels).

Table 4.3.1. Invertebrate species diversity per site, measured by Margalef's Index. 'Abundance' refers to the total number of specimens recorded from that site. Experimental habitats were bulldozed after the first sample date.

Site	1993		1994	
	Abundance	Score	Abundance	Score
1 Ohau	272	2.85	751	3.32
2	1412	3.86	879	3.54
3	2620	6.23	1643	7.16
10 Rec Trt	2220	3.38	2734	5.69
11	2601	3.69	4072	4.09
12	4608	5.17	1935	6.87
13 Expt Sh	28060	2.64	6250	5.95
14	2872	3.77	2287	5.30
15	1339	4.58	4182	3.72
19 Cont Sh	4040	3.61	2704	4.56
20	41300	2.73	3724	4.01
21	17707	3.17	1219	3.66
16 Expt W	4735	6.26	3175	6.45
17	1521	6.55	3600	6.11
18	3652	5.61	4925	6.47
22 Cont W	980	7.70	359	7.48
23	9232	4.16	886	6.48
24	10571	5.07	863	6.95

Table 4.3.2. Invertebrate species diversity per habitat, measured by Margalef's index. 'Abundance' refers to the total number of specimens recorded from that site. Experimental habitats were bulldozed after the first sample date.

Habitat	1993		1994	
	Abundance	Score	Abundance	Score
Ohau	4304	7.67	3273	8.15
Rec Trt	8889	6.82	8744	9.14
Expt Sh	32271	5.01	12719	8.78
Cont Sh	63047	5.15	7647	6.48
Expt W	9926	9.34	11700	9.82
Cont W	20783	7.75	2108	10.19

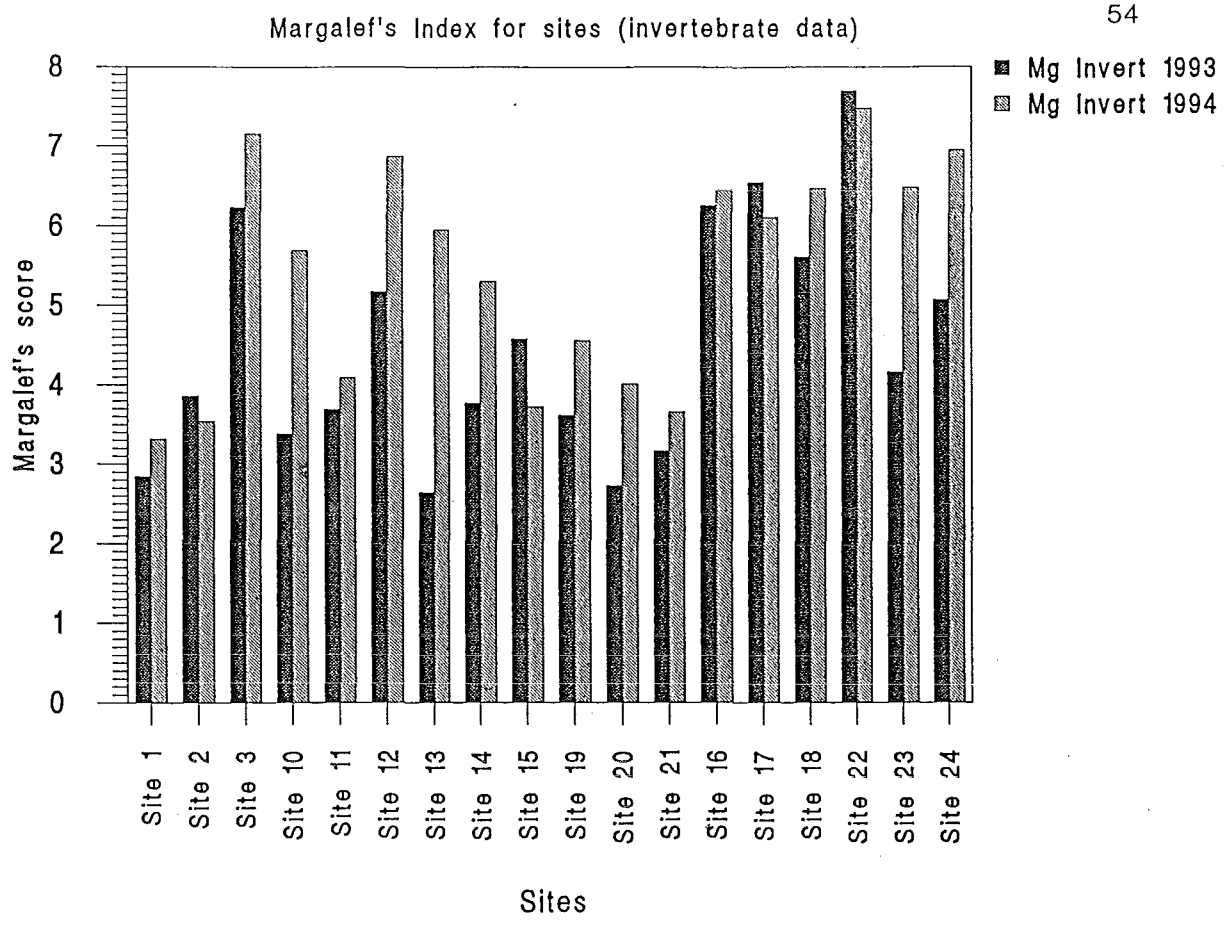

Figure 4.3.1. Invertebrate species diversity per site, measured by Margalef's Index. Experimental sites were bulldozed after the first sample. Sites 1-3 are from the Ohau habitat (flooded after first sample); 10-12 from Recovering Treatment area (bulldozed 1992); 13-15 from Experimental Shingle habitat; 16-18 from Experimental Willow; 19-21 from Control Shingle (undisturbed); 22-24 from Control Willow (undisturbed).

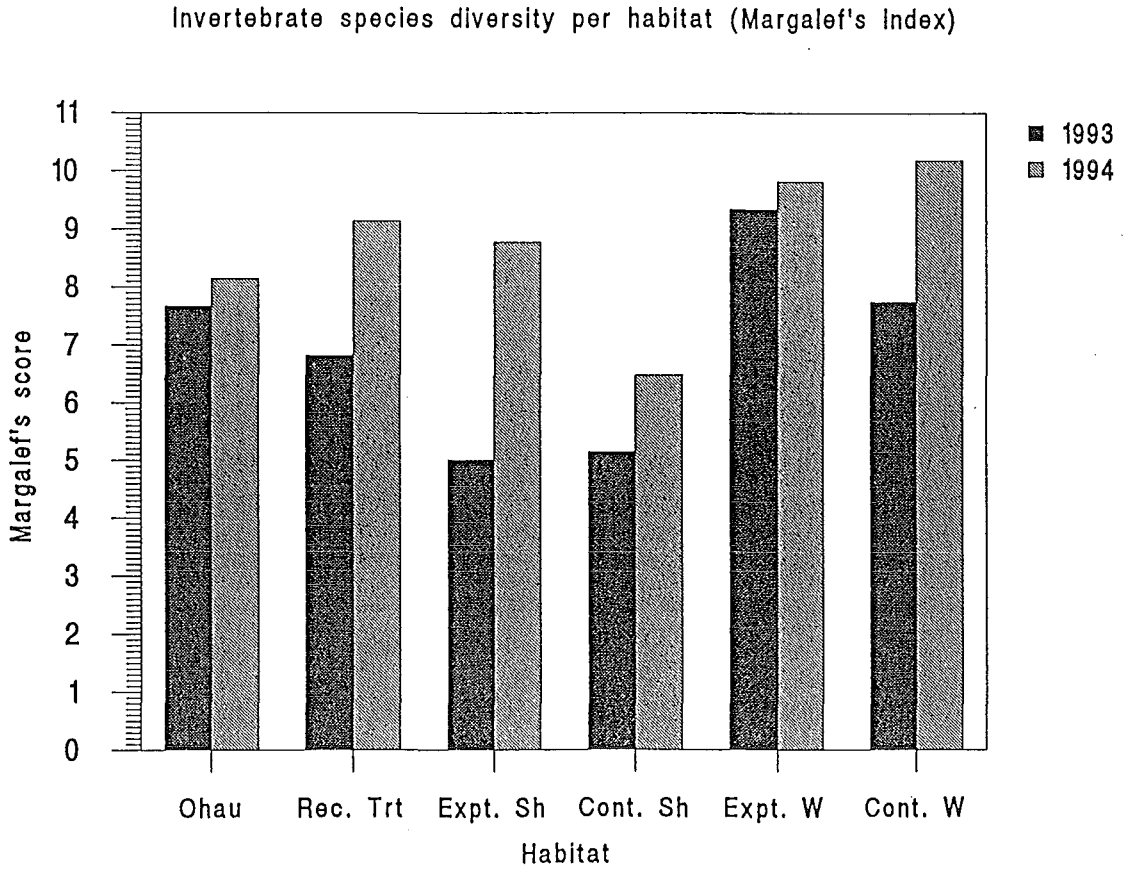

Figure 4.3.2. Invertebrate species diversity per habitat, measured by Margalef's Index. (See Table 4.1.1a for habitat labels).

4.4. Relative distribution of invertebrate taxa amongst five orders per habitat

Table 4.4.1 summarises the data shown in Figures 4.4.1a, b and c. The Ohau samples were more or less equivalent in their distribution of taxa throughout the 5 main orders although the 1993 sample had proportionally more miscellaneous taxa. Similarly with the Recovering Treatment habitat: although the number of taxa was higher in 1994, they occurred roughly in the same proportion as the 1993 sample. Slightly more Coleoptera and Diptera were recorded than from the other orders.

In 1993, the Experimental Shingle habitat had fewer Coleoptera taxa and proportionally more miscellaneous taxa than either the Ohau and Recovering Treatment area. However, after bulldozing the distribution of taxa was comparable to the Recovering Treatment habitat. Similar to the pre-treatment Experimental Shingle habitat, fewer Coleoptera and more miscellaneous taxa were recorded from the Control Shingle habitat than for either the Ohau and Recovering Treatment area. However, the Control Shingle habitat maintained a consistent distribution of these taxa amongst the five main orders from 1993 to 1994.

A higher number of Hymenoptera was recorded from the Experimental Willow habitat in 1993 than in any other habitat, but they decreased in number after bulldozing so that the proportions of taxa across the 5 major orders resembled that of the Recovering Treatment area. The 1993 sample of the Control Willow habitat contained fewer Hymenoptera taxa than the Experimental Willow habitat, but maintained a fairly consistent distribution of taxa across the orders between sample dates.

4.5. Temporal stability (persistence) of invertebrate and plant taxa per habitat

Invertebrates

The taxonomic assemblages in every site underwent some changes in species composition, even in the Control habitats. Table 4.5.1 and Figure 4.5.1 summarise the changes in invertebrate taxa between the 1993 and 1994 samples. The

Experimental Willow habitat had the highest possible species richness and Control Shingle the lowest. However, after bulldozing the Experimental Willow habitat lost the most species from its original species composition while (surprisingly) the Experimental Shingle lost the least amount. However, the Experimental Willow and the Recovering Treatment habitats gained the most species while the Control Shingle gained the least. As expected, both the Control Shingle and the Control Willow habitats shared a higher percentage of their invertebrate taxa between the two sample dates than the other habitats.

Plants

Plant taxa showed a similar pattern of species turnover to that shown by the invertebrates (Table 4.5.2 and Figure 4.5.1). Again, the Experimental Willow habitat had the highest number of taxa when the totals from both dates were combined, and the Control Shingle had the least (although the Control Willow habitat also had a low combined richness score).

In contrast with the invertebrate data, the Ohau habitat recorded the greatest loss of plant species (after flooding) while the Control Willow and the Control Shingle lost the least. Both the Experimental Shingle and Willow habitats gained the most species after bulldozing, and the Control Willow habitat gained the least. As expected, the Control habitats shared a higher percentage of their species between sample dates than the other habitats.

Table 4.4.1. Number of invertebrate taxa per habitat from five orders. Habitats are listed at the left of the table (see Table 4.1.1a for labels) while five invertebrate orders are across the top. Numbers within the table are the number of species in each order. 'Miscellaneous' refers to taxa not included in the five main orders. Experimental habitats were bulldozed after the first sample.

	COLEOPTERA	DIPTERA	HYMENOPTERA	HEMIPTERA	ARANEAE	MISCELLANEOUS	TOTAL TAXA PER HABITAT
Ohau 1993	13	8	10	8	9	18	66
Ohau 1994	9	14	14	9	7	14	67
Rec Trt 1993	14	11	8	9	8	13	63
Rec Trt 1994	21	16	9	13	10	15	84
Expt Sh 1993	1	9	11	6	8	18	53
Expt Sh 1994	14	14	14	10	13	19	84
Cont Sh 1993	6	7	11	6	10	18	58
Cont Sh 1994	2	9	9	10	12	17	59
Expt W 1993	9	14	26	9	9	20	87
Expt W 1994	22	20	13	11	9	18	93
Cont W 1993	8	13	14	9	11	23	78
Cont W 1994	6	21	16	8	10	18	79

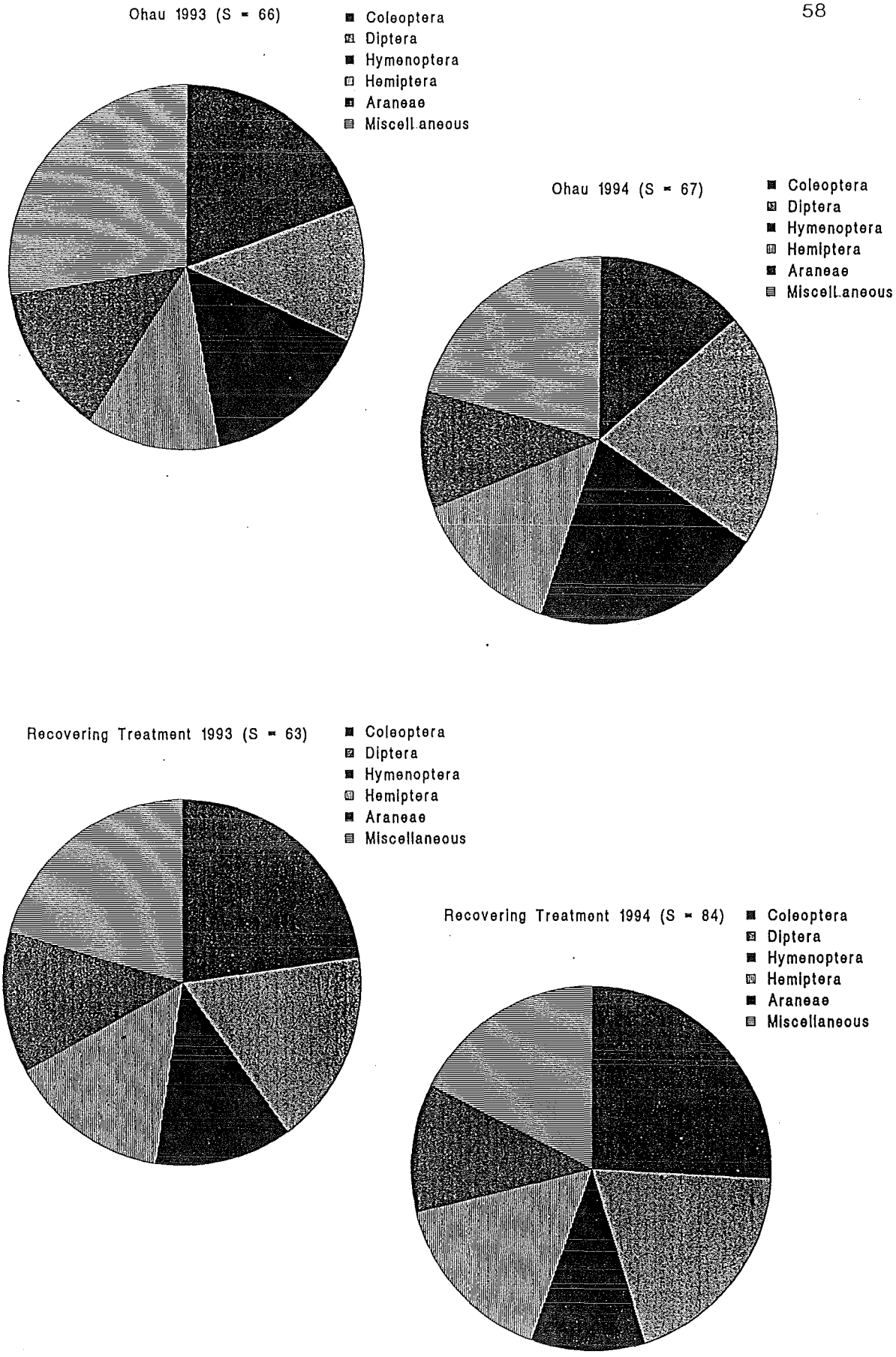

Figure 4.4.1a. Relative number of invertebrates in the Ohau and Recovering Treatment areas from two sample dates. 'Miscellaneous' refers to taxa which are not from the five major orders.

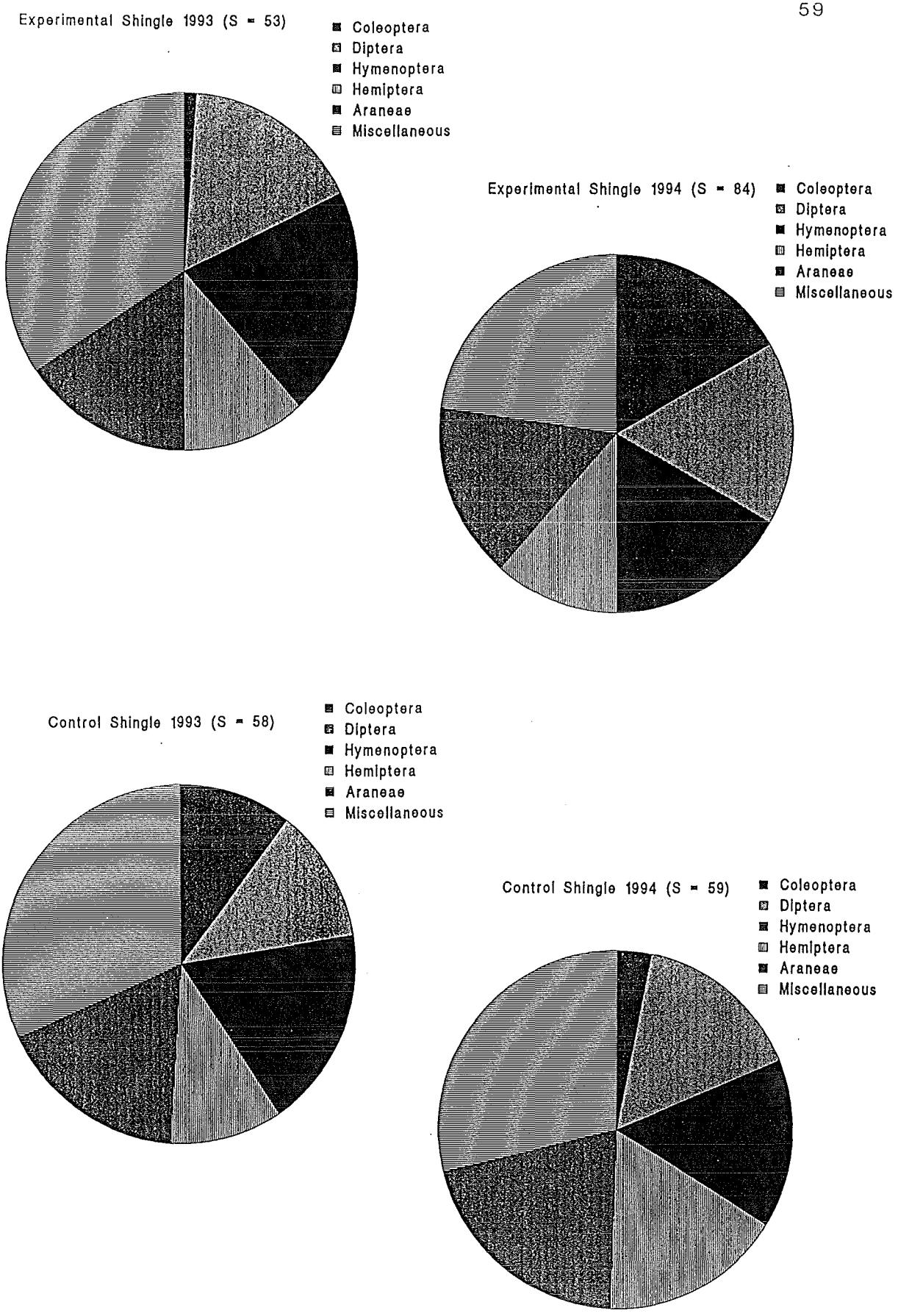

Figure 4.4.1b. Relative number of invertebrates in the Experimental Shingle and Control Shingle habitats from two sample dates. The Experimental habitat was bulldozed after the first sample. 'Miscellaneous' refers to taxa which are not from the five major orders.

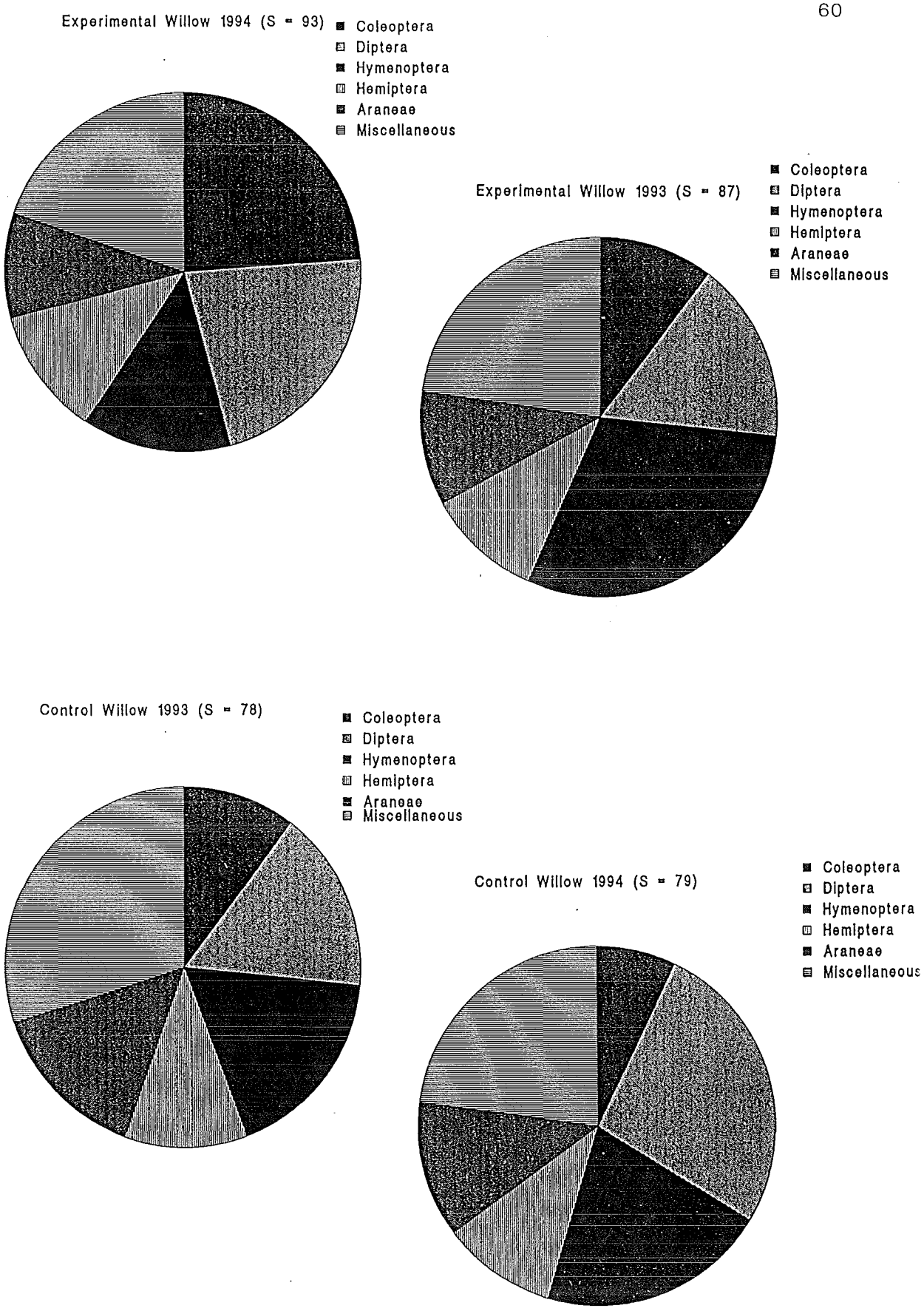

Figure 4.4.1c. Relative number of invertebrates in the Experimental Willow and Control Willow habitats from two sample dates. The Experimental habitat was bulldozed after the first sample. 'Miscellaneous' refers to taxa which are not from the five major orders.

Table 4.5.1. Temporal stability (persistence) of invertebrate taxa per habitat between sample dates. Numbers are the number of taxa in each category. (See Table 4.1.1a for habitat labels).

	Ohau	Rec Trt	Expt Sh	Cont Sh	Expt W	Cont W
Total number of species found over both dates	97	108	97	82	132	109
Number of species lost (not found in 1994 sample)	30	24	14	23	39	30
Number of species gained (found in 1994 but not in the 1993 sample)	31	45	44	24	46	31
Number of species shared between sample dates	36	39	39	35	47	48
Percent of species shared	37.11	36.11	40.21	42.68	35.61	44.04

Table 4.5.2. Temporal stability (persistence) of plant taxa per habitat between sample dates. Numbers are the number of taxa in each category. (See Table 4.1.1a for habitat labels).

	Ohau	Rec Trt	Expt Sh	Cont Sh	Expt W	Cont W
Total number of species found over both dates	74	73	69	46	77	42
Number of species lost (not found in 1994 sample)	34	12	12	7	16	3
Number of species gained (found in 1994 but not in the 1993 sample)	11	15	21	11	18	5
Number of species shared between sample dates	29	46	36	28	43	34
Percent of species shared	39.19	63.01	52.17	60.86	55.84	80.95

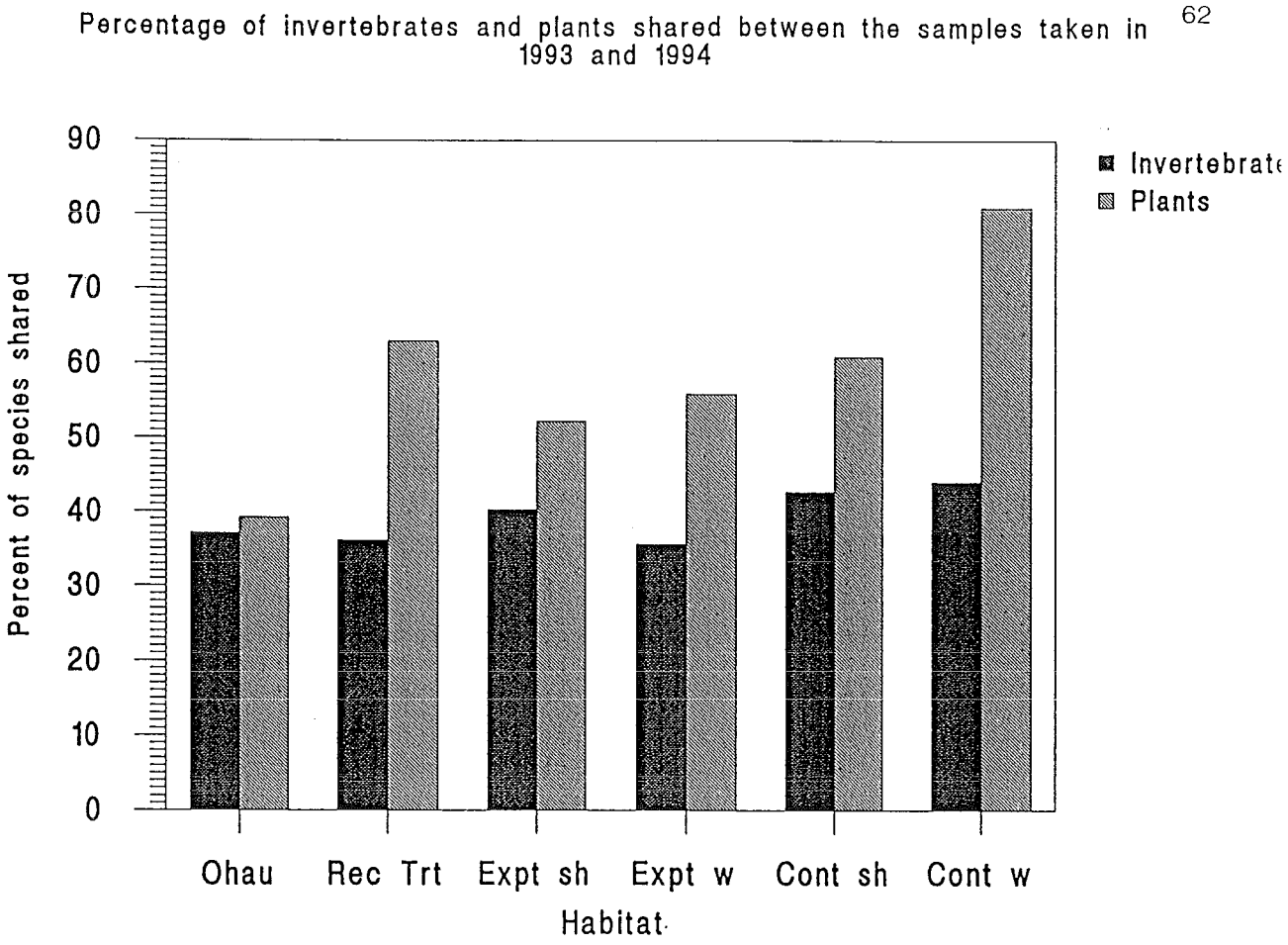

Figure 4.5.1. Temporal stability (persistence) of plant and invertebrate taxa per habitat between sample dates. (See Table 4.1.1a for habitat labels).

4.6. Environmental data per habitat

Willow habitats had more litter than the other habitats (Table 4.6.1 and Figure 4.6.1) and a slightly higher % cover of vascular plants. Shingle habitats had a higher % cover of lichens (primarily rock lichens) and bryophytes than other habitats (although the Bryophyte cover was greater in the Control Shingle habitat). The variable % cover of rabbit droppings contributed little to the overall total cover score.

The Control Willow and Shingle habitats remained relatively constant for all environmental variables measured from 1993 to 1994, while a decrease in plant cover was recorded from the Ohau habitat as a result of flooding (Figure 4.6.2). There was an increase in total % ground cover in the Recovering Treatment habitat due to an increase in vascular plants and litter. Correspondingly, there was a decrease in the amount of bare ground exposed (hence a decrease in exposed silt and sand). After bulldozing, as expected there was a loss of vegetation from the Experimental habitats. Litter and lichens decreased in the Experimental Willow and Shingle habitats respectively. The amount of bare ground exposed and some of its components increased accordingly. Loss of willow trees from the Experimental Willow habitat also led to a reduction in shade and litter depth (not shown).

HABITAT	% TOT COVER	% COV VEG	VASC	LICH	BRYO	% COV LITTER
OHAU 93	31.2	30.12767	29.84717	0	0.280667	0.819333
REC TRT 93	20.9	17.50833	17.475	0	0.016667	2.991667
EXPT SH 93	40.93333	36.59983	18.2075	15.05567	3.387	2.833167
CONT SH 93	50.75	45.28617	18.85983	12.182	14.2445	3.4695
EXPT W 93	86.11667	51.4	49.6	0	1.8	34.28333
CONT W 93	93.68333	60.01667	58.65	0.1	1.266667	32.96667
OHAU 94	11.4	10.386	10.38333	0	0.002833	0.827667
REC TRT 94	33.26667	27.26667	27.23333	0.016667	0.016667	5.733333
EXPT SH 94	31.25	26.41117	24.27783	0.858333	1.275	4.0695
CONT SH 94	59.16667	51.961	22.82867	13.62867	15.50367	5.194333
EXPT W 94	38.33333	34.29167	34.29167	0	0	3.608333
CONT W 94	96.01667	66.58333	65.01667	0.216667	1.35	28.58333

% COV RABBIT	% BARE GROUND	SILT	SAND	GRAVEL	PEBBLE	COBBLE
0.252667	68.71667	2.141667	2.583333	6.641667	36.98333	20.2
0.416667	79.08333	45.4445	9.8195	4.400333	11.90583	7.119667
1.499833	59.06667	9.530667	10.832	5.3195	23.79733	9.487667
1.9945	49.25	35.62783	0.354167	3.2945	5.616667	4.523667
0.433333	13.88333	5.811167	6.375	0.191667	1.0445	0.461167
0.733333	6.316667	4.902833	0.8445	0.027833	0.275	0.266667
0.186	88.6	0.083333	2.683333	9.783333	53.31667	22.48333
0.266667	66.73333	33.65833	3.7195	7.389167	10.6975	10.51967
0.7695	68.75	25.0445	6.45	5.364	16.80567	14.67783
1.877667	40.83333	25.846	0.8875	3.492	6.481	4.100333
0.433333	61.66667	39.2945	2.565333	2.807333	10.28783	6.712833
0.85	3.983333	3.033333	0.525	0.041667	0.075	0.308333

BOULDER	LIT DEPTH	SHADE	# WILLOW <5m	# WILLOW >5m
0.166667	0.45	0	0	0
0.3945	1.15	0	0	0
0.016667	1.4	0.029167	1.133333	1.533333
0	0.941667	0	0.333333	2.266667
0	2.75	0.7875	2.933333	1.333333
0	2.966667	0.770833	2	1.666667
0.25	0.575	0	0	0
0.75	1.716667	0	0	0
0.491667	1.625	0	0	0
0.027833	1.333333	0	0.333333	2.266667
0	2	0	0	0
0	3.016667	0.775	2	1.666667

Table 4.6.1. Environmental data for each habitat from two sample dates. (See Table 4.1.1a for habitat labels). The environmental codes refer to the following variables:
% Total ground cover (sum of the cover from vegetation, rabbit droppings and litter); % vegetation cover (sum of vegetative components); % cover of vascular plants; lichens; bryophytes; % cover of litter; % cover of rabbit droppings; % bare ground exposed (sum of bare ground components); bare ground components (silt, sand, gravel, pebble, cobble, boulder); litter depth; shade; number of willow trees closer than 5 metres; number of willow trees between 5 and 30 metres away.

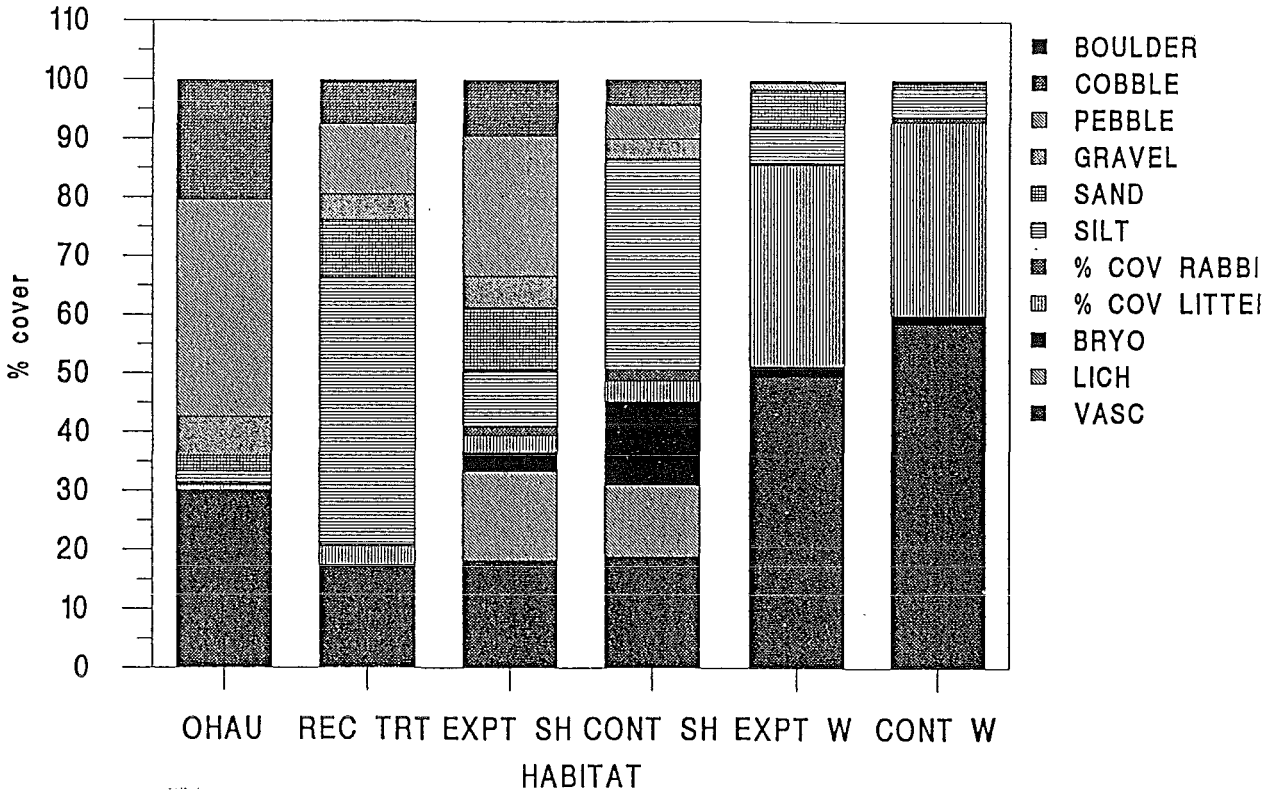

Figure 4.6.1. Environmental data for each habitat in 1993. The total % of ground cover can be obtained by summing the variables % cover rabbit droppings, % cover of litter, plus the three vegetation categories. The total amount of bare ground exposed can be obtained by summing the substrate components.

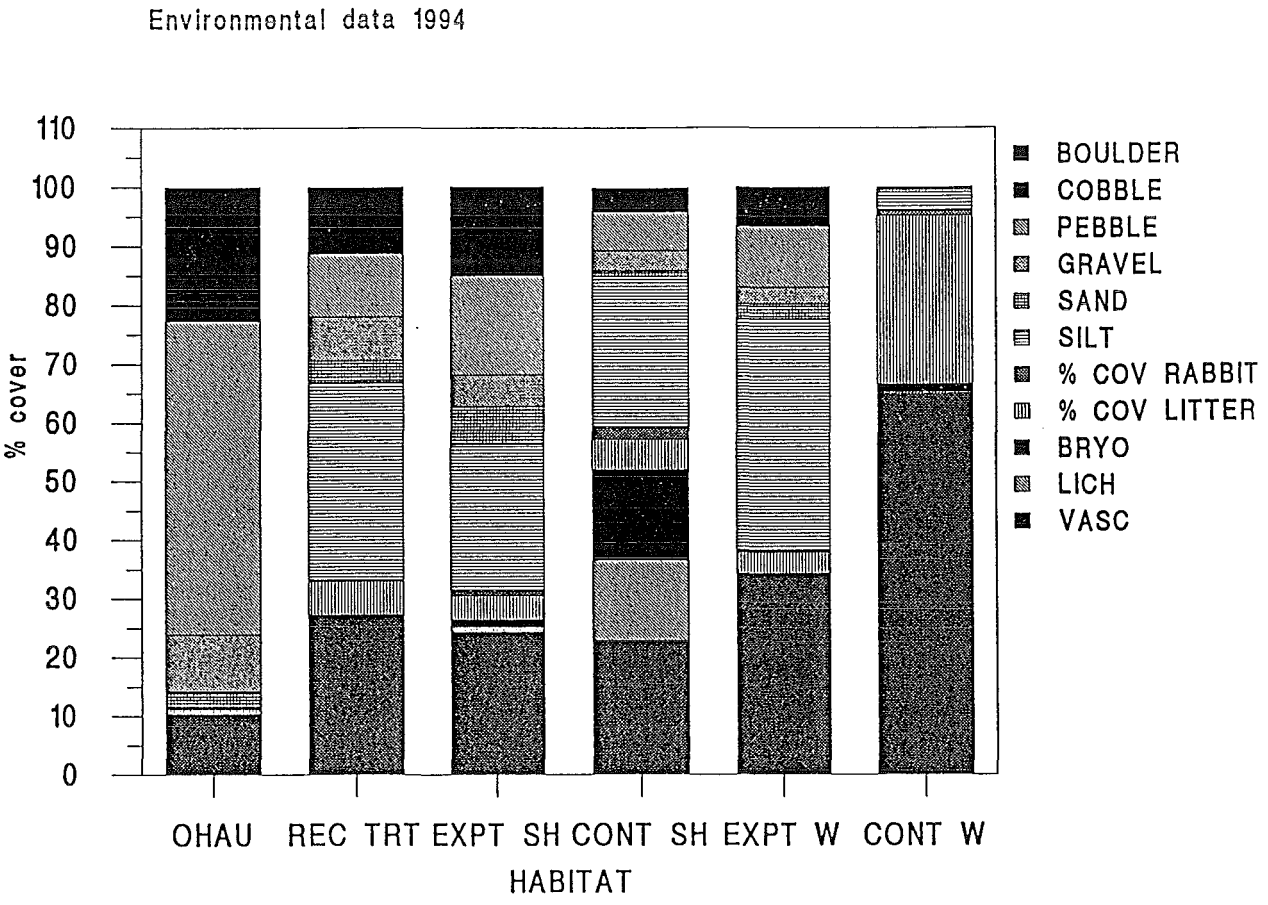

Figure 4.6.2. Environmental data for each habitat in 1994. (Experimental habitats have been bulldozed; Ohau habitat has been flooded; Control habitats were undisturbed).

ANALYSIS OF COMMUNITY DATA WITH COMPLEX STATISTICAL METHODS

4.7. Community classification

4.7.1. Classification of invertebrate data

Seven groups were distinguished (Figure 4.7.1 and Table 4.1.1a for site codes). The range of dissimilarity values was from 0.27 to 0.61. The first division separated the Ohau (both samples of site 1 (site label 2-1 and 7-1) and 1994 sample of site 2 (7-2)) from the rest of the sites. (Site 2 was split from site 1 in a later division, but were grouped together for convenience). This represented the first group. The second division separated all the 'bulldozed/disturbed' sites from the undisturbed sites.

Within the bulldozed/disturbed arm of this division, three groups were recognised. The 1993 sample of site 3 (site label 2-3; Ohau) was assigned to group 2. Group 3 contained all the 1993 Recovering Treatment samples (2-10, 2-11 and 2-12) plus a 1994 sample of site 11 (7-11), and two bulldozed Experimental Shingle sites (7-14 and 7-15). Group 4 contained 1994 samples only, of site 3 (Ohau) and the remaining Recovering Treatment sites (7-10 and 7-12), the remaining bulldozed shingle site (7-13) and all bulldozed Willow sites (7-16, 7-17 and 7-18).

Within the undisturbed arm, three groups were distinguished. Group 5 consisted of all the pre-treatment Experimental Shingle sites (2-13 to 2-15) and the Control Shingle sites (sites 19, 20 and 21 from both dates). The next two groups contained the 'Willow' sites; group 6 included all the 1993 samples from the Control (2-22, 2-23 and 2-24) plus all the pre-clearance Experimental sites (2-16, 2-17 and 2-18). (The Control Willow sites 23 and 24 had the lowest dissimilarity score in this classification). Group 7 consisted of all the 1994 samples from the Control Willow habitat (7-22, 7-23 and 7-24).

The invertebrate species by site (two-way) table is shown in Table 4.7.1. The majority of taxa occurred in more than one site and those species which were

exclusive to single sites were not abundant. Distinctions between 'Shingle', 'Willow' and ' bulldozed/disturbed' sites were not apparent through presence or absence of guilds of species but rather from differing abundances of ubiquitous species. For example, the Hemipteran species 6 (*Nysius huttoni*) occurred in higher abundances in the bulldozed sites than in the Willow and Shingle sites respectively. Similarly with the introduced Thrips (Thysanoptera, species identification number 94) and aphids (Aphididae, 36). The harvestman (*Phalangium opilio*, 21) occurred indiscriminately in almost every site. Cecidomyiidae (135) were more abundant in Willow areas while ants (*Monomorium antarcticum*, 11) were in higher numbers in Shingle sites. Individual taxa which characterised habitat types were more apparent in the species ordination diagrams (see later).

4.7.2. Classification of plant data

Seven groups of sites could also be distinguished in the plant classification (Figure 4.7.2 and Table 4.1.1a for site codes). Like the invertebrate classification, the first division separated some Ohau sites from the main body. Site 1 of the Ohau (both dates) had a dissimilarity score of 1 from all the other sites, and a score of 0 between sample dates (both samples for this site contained no plant taxa). The second and third divisions separated the Ohau site 2 (both dates) and Ohau site 3 (both dates) from the remaining sites.

Unlike the invertebrate classification, the next main division did not separate the bulldozed from the undisturbed sites. Rather, the Shingle sites were separated from the Willow and Experimental sites. Group 4 contained all pre-treatment Experimental Shingle (2-13 to 2-15) plus the Control Shingle sites (both dates). The remaining 'bulldozed and willow' arm was split into three groups (5, 6 and 7). Group 5 contained all the bulldozed Shingle sites (7-13 to 7-15) and two of the bulldozed Willow sites (7-16 and 7-18). Group 6 contained all the Recovering Treatment sites (both dates) plus one pre-bulldozed willow site (2-16). Group 7 contained the two remaining pre-treatment Experimental Willow sites (2-17 an 2-18) plus the all Control Willow sites (both dates).

The plant species by site matrix is shown in Table 4.7.2. Similar to the

invertebrate data, the majority of species occurred in more than one site and those species which were site specific occurred in low densities. The Ohau sites were distinctive; Ohau site 1 had no plant species; Ohau site 2 could be separated on the presence of bare shingle native colonisers such as *Epilobium melanocaulon* while shingle sites had a predominance of lichens (especially rock lichens), bryophytes, mat plants (*Raoulia* spp.) and *Muehlenbeckia axillaris* (although this species also occurred in sward in the Experimental Willow sites). Pasture grasses such as *Anthoxanthum odoratum* and *Agrostis capillaris* occurred primarily in willow sites and site 3 from the Ohau. Recovering Treatment sites had a characteristic presence of willow (*Salix fragilis*) stems regenerating from broken fragments.

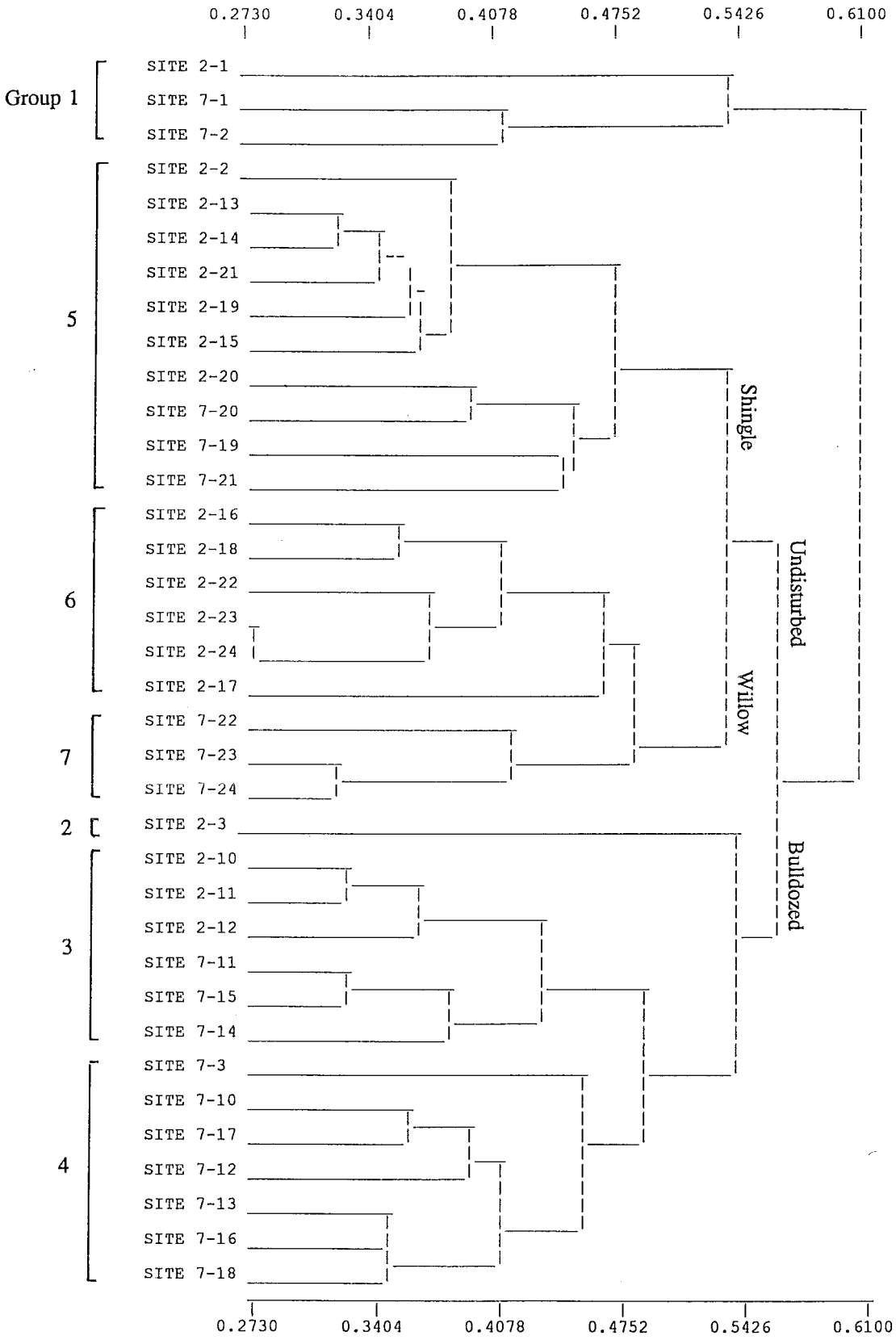

Figure 4.7.1. Invertebrate site classification dendrogram from $\log_{10}$ transformed data. Numbers

Table 4.7.1. Invertebrate site by species two-way table. Species codes are across the top (read vertically) and site codes are at the left. Numbers in the body of the table are the 'global' abundance categories generated by the program (1-9 for species abundance, . = absent). Species codes are in Appendices; site codes are in Table 4.1.1a.

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          2211111223232623333323123212522332312322222222222222212132111192225112112231112312473413221322227291122118 18 12121 27227112
          2929608500313840001061841597905004097174282585876198753142700815240702917397640913697629871989943247059505876691089030714 2
          9335362584042 46383210659410 377925325783202903537001802027 9  514 70 7928 48 1121    8 56 14862 8 554660 9  29740   99 606

SITE 2-1  ..........................................................................1.....................1........1....1.....1...
SITE 7-1  ..........................................1.............1.....1...........1.............................2...
SITE 7-2  ..........................................1.............1..1.......1.........2..1.....................2...
          ------------------------------------------------------------------------------------------------------------------------
SITE 2-2  ..1.....................................................1......1........111.........1..........1.....1.....111.
SITE 2-13 ..........................................1.........1......1.....................11.....211.....1...1.1.
SITE 2-14 ...1.....................................1.................1.1.2.....................2....2.1...21..2.......1.2.
SITE 2-21 1..1.....................1.................2..........1...1.....1.1....1.............3....2....12............1..
SITE 2-19 ..................1.......1.....................1....1.1..........1.1..2....1.................1.......1.......1.....
SITE 2-15 .1..1.....................1.....................1...1..1.......2.............2..1.2.11..21..2........11.
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SITE 7-19 ..........................................................................1...11.............1...1..11..1.......2...1...
SITE 7-21 ..........................................2.....................1.....................2.......2..3.1.............
          ------------------------------------------------------------------------------------------------------------------------
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SITE 2-17 ..........................................................1..1.............1.........12.1.1.11..2...1.......1...
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SITE 7-23 ..........................................................2.1.1.12.1............1....1......1.1..21..........131.
SITE 7-24 ...2.....................................2.1......11....2.1..12.1..........1..11.1.1...1.1.1.............11..
          ------------------------------------------------------------------------------------------------------------------------
SITE 2-3  ..........................................................................1.....................1.......1.14.1.....21..1...
          ------------------------------------------------------------------------------------------------------------------------
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SITE 2-11 .............1.....................................1....1............1..11.1.2.............2..1.........1...11..
SITE 2-12 ....2.1...........1.................1.................2..........12......1.......1.12.2.........11..2.1.21..........1.22..
SITE 7-11 ......................1.....................................1.......111.............2.1...........2.2.....2.......3...12.1
SITE 7-15 .1.1.....................................................1...1.2.....11.....................1...11.............2..121..
SITE 7-14 ...12...1.................1.....................1.........1..1...........1.1.........2..2.2...2141......11..1.1.
          ------------------------------------------------------------------------------------------------------------------------
SITE 7-3  ..........................................................................2.........1.............1.....2.22.....221111.13
SITE 7-10 ..............1.....................................1.....................1..1.............11.1..211...1.411.221.
SITE 7-17 .............1........2.........2.................1...1.....................1....1.............1.1.1...1........1..1...1.
SITE 7-12 ....11...1.2...1.1...1.........11.1.................1.......3......1.........1.....................1..11.................12..
SITE 7-13 ..1.....................1...1........1.....................1.......1.....1.....1..11.............11.......2.1...1..1..1.1.
SITE 7-16 1.........1.1.....................................1.1....1........1..1..1...........1.........1..1.....1.3........2..1221
SITE 7-18 ...1...1.................2....1.....................1.................1.....1..1.1.1..........1..22.21.........221..11.
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Table 4.7.1. cont...

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          4211122112222212221741111116211112611211111212121512612612122332212783121829992112819116 221321811216111223332
          1335842987767676017229065826056325374458854501101503731434214769828955261925341781638198568170304453521569911116
           196 38849173548506   44315 28066  9991354432926  57 53 364    1495   027 1     83  0 09   83061 3537 121796786

SITE 2-1  ..1.............24.....................2...1.....4.....3.....233..3.....1.....
SITE 7-1  .....3..1.....1....2.....3.....2...3...1631141.1..1.2.1.....
SITE 7-2  ..2.....1....1....1.....1.4....2.....2.31.....13.....53.15.....1.2.....
----------------------------------------------------------------------------------------------------------------------------
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SITE 2-13 .....1.....2.3....1.....2.12.....2..11...159..41.....12.....
SITE 2-14 .....2.....11.....12.....1.3.....2.....12..1....157..42.....22.....
SITE 2-21 .....1.....11....21.....2.2.....2.1.....12..11...168..4.....2.....
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SITE 2-15 .....1.....1...1.....1...12.....3.1.....2..2....156..51.....13.....
SITE 2-20 .....1....3.2.....1...22.....11....2.....31.....1...159..5.1....3.....
SITE 7-20 .....1....13.3.....1....2.....21.....21.....21.2.....1..154..7.....31.....
SITE 7-19 ..1.....1....12....1..1..1..1....2.....211....1.....2112.....11.1.....54..7.1...122.....
SITE 7-21 ..1.....1...2.1..1.....1..1.1.....2.....2.....1.11.....2.11....264..5.....1.....
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SITE 2-16 1.....1.1....1.2.1..12.1.....111..11..11....3.22.....21..121..267..422.....21.....
SITE 2-18 1...11.....1.1....1....1113....21.....11.....2.....3.32.....2...212..157..4331.....1.....
SITE 2-22 .....1.....1.....1..1..22.2..1.1....1121.....3.....3.2.....21..122..156..342....111.....
SITE 2-23 .....1.....1..1..22.2.....1312..1.3.....3.21.....2....2...248..322....221.....
SITE 2-24 ...1.1.....2.....1.12212..2.1.....2.2....2.....3111.....2...121..158..422....231.....
SITE 2-17 1...2.1.....1.....1.21..2.2....1....24..3122.22....2..2..1..2....11..156..3421...141..1....1.....1
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SITE 7-22 .1....1..1.....2..2.1.1.2..1.4.....1211....21....2.32.2.1.2.....1..241..3321....31.....
SITE 7-23 .1.....1.....1...2..2.12..3....4.....123....21....3.1.11..11.....1..153..432...1.31.....
SITE 7-24 .....1.2.....1..1..31.2..1.4.....222....1.....3..11....11...1...253..423....131.....
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SITE 2-3  ...1.....1....1.21.11...3.1.1..2..2...1...2..2...3..2..1..221..2.22356..44.2..113....12.1.11111.....
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SITE 2-10 .....1....1....3.....1.....1..13.....13...1...265..5.....124.....
SITE 2-11 .....1....1.1....1.....1.....1.....3..1....13.....364..5..1....34.....
SITE 2-12 1.....1.....12.1.12.....1....13.21.....12...1...266..622....44.....
SITE 7-11 ..1.....12.....1..2.....2.....1.1.....3.23.....13.1.....371..61.....45.....
SITE 7-15 .....11...1.....1.....111.....3.23.....22.....262..7.....43.....
SITE 7-14 .....2.1..2.....1..21.2....2.....12....1.....3.22.....1.....264..6.1....43.....
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SITE 7-3  .....1.12..1.....112..11.....1....1....2.1..13.33...23133..2...363113.33..245..1.....21111.
SITE 7-10 .....1....11.122.....1.412.13.....31..2.11363.15123.4446.....
SITE 7-17 ...1....1.....2111.111..1...2....11.....1.212.3.33.....21..1.21274..414413325.....
SITE 7-12 .....22.....13...1.1...2.....1.....32111.13..1..121..3.21255..424422346.....
SITE 7-13 ..1.....2.....1.111..112.....21...22...1.....11.23.23.....12...2...167..512211136.....
SITE 7-16 .3..1.2.....1211.11.2....2.....1.2....2....13.23.....11...2...365..6222112461.....
SITE 7-18 .....1..1.....1...21....2.12.1.1....321.1.....1..2113.24.....1...3.11366..52222.347.....
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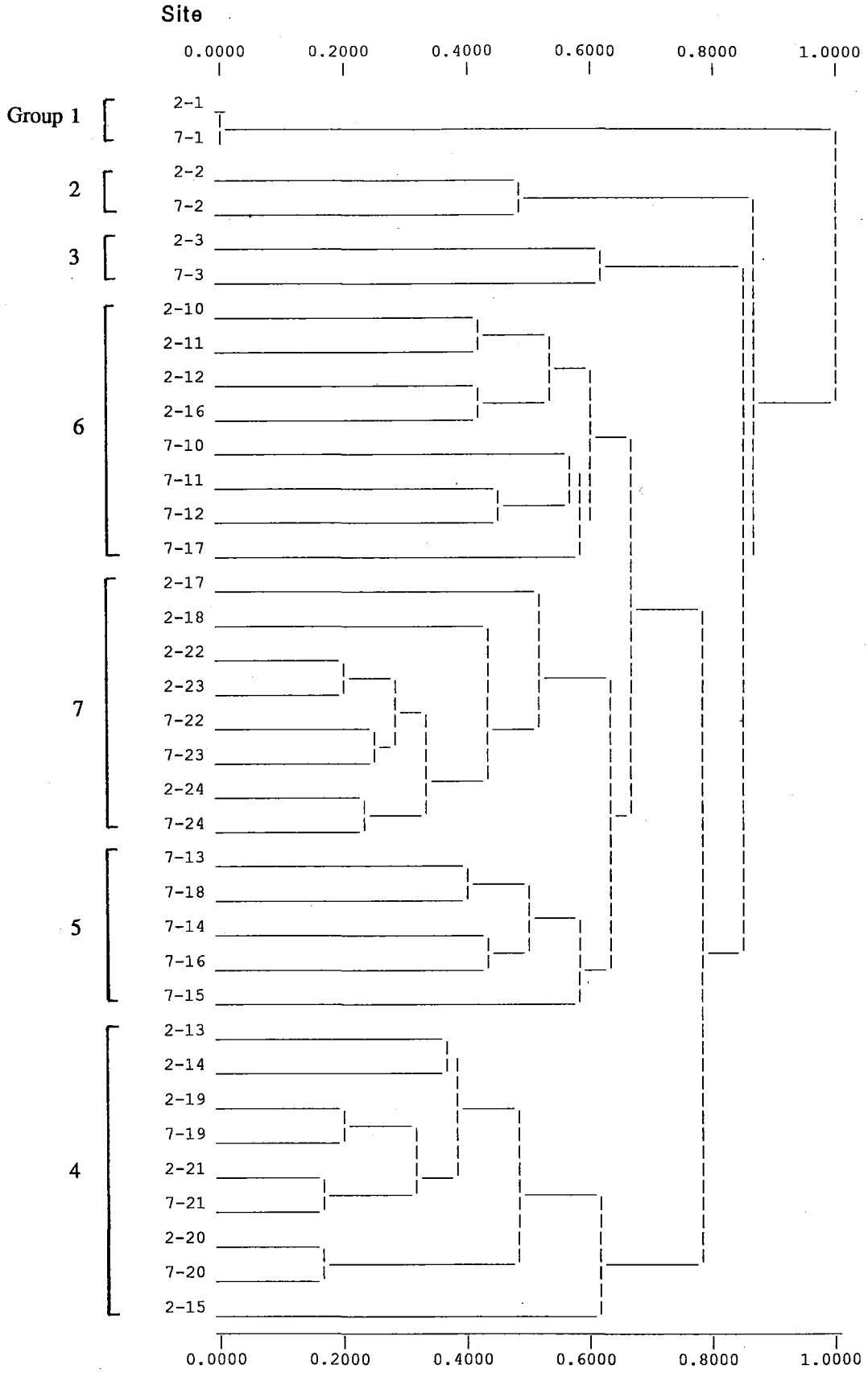

Figure 4.7.2. Plant site classification dendrogram from $\log_e$ transformed data. Numbers across the top and bottom of the diagram are dissimilarity values. Hyphenated numbers at the left represent site codes (See Table 4.1.1a for site labels).

Table 4.7.2. Plant site by species table. Species codes are across the top (read vertically) and site codes are at the left. Numbers in the body of the table are the 'global' abundance categories generated by the program (1-9 for species abundance, . = absent). Species codes are in Appendices; site codes are in Table 4.1.1a).

```
      291622761379664333383471132581811235132229662361885133775138156421241883281111133511144196134224551357625335117711411
      090010973261400505200210050482612146194619121 352720  795311256802507401391132503220117216839303111835822124817724
      8 205    4   1072 1 0  5434  8 153  7   4    0 3   36       0     2 3       68072 16   9  2     0 3         95   4  1

2-1   ..........................................................................................................................
7-1   ..........................................................................................................................
      ==========================================================================================================================
2-2   .......1.............1.............................1.........1.....................42.1.43..11..1...........31.1.
7-2   ............1.......1..2.....................4.............1..1.................1....61.1..2.11...1.........1.3..2.
      ==========================================================================================================================
2-3   .......1..........1....1..2.1..1...1.11..1.5.1...21.131211.123312..1132.221111...131212.147.15.21312.841.5714..22..1
7-3   ....................1.......................3..1....1...1.111....111..11....15...11.1..22.35..11.1.22..5412..12..1
      ==========================================================================================================================
2-10  ....1.1.............11.......1..1....1..1.1..........1.1.111111.11.121.1.........1.21.1.3..1311..12223111.........
2-11  ................1..1...1...11.11.13.1....1...1...1.1.....111.11.1...24.1.........1.1..1.2...4....122232..........
2-12  .....1.........1...11....1......1..1..1..1....21..1..11.11.122212.1..122.1.........111..1.7..15.11.12552211.........
2-16  .......1.........1.1.......1...1..1....1....3.....1..1....1111111......1............1.3..1.7.2341..1223554.1.........
7-10  .......1......1.......2..1...12..1...1....3..212..1.111....2122111.1..12..1.........2.1..2.3..13.1..1222.14..........
7-11  .......1..............3..............5....2..11.1...1.1......131.2.1...22...........2.21.411.115.11.1545151.........
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      ==========================================================================================================================
2-17  .51....12..........2...11..1..1...2..11....2........1....2....51...1.12..1.........1271.2.6.1255...2338.6.3.........
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2-22  ..................................21...1...18........1.....3...111.....1............1.3..2...2121...5236.5.3..1.......
2-23  ....................1.................1.......9........3.....4..1.11.....11.................2...2122...6128.6.3..........
7-22  ..............1.....................211..1...16..............5...121.....2.1..........1.3..1...7221...6357.6.2..1.......
7-23  ..................1..................2..1....5........11....4..1.11.....1.1..........1....3.2.7.21...6139.7.4.........
2-24  ......................1..........1111.......7.1......1.1...3..1.1......1.1..........1.8.16..12114...4236.4.3..11.....
7-24  ..............1.......1...........1.1.21....5.1......1.2...4.11.1........1..........1.8.13...6225...4237.6.4..11.....
      ==========================================================================================================================
7-13  ......1..................12.1.11.1........1...211...2.21.....22.11.2...21............2.41.411.314.1..2214.1.1.........
7-18  ...1...2............111...1..111.1...1...111.12.1...1.31...1121111.2.1132.1.........2.4..6.4.226....333713.2.........
7-14  ....1..26....11.......2.......11.......1..2...2...2..2.2...1.22.211....311.........1.8.25..22124...52.5...3..32..2..
7-16  ......11..............1...11.211......1..11..2.....1.11.....1213..1.1.711..........1.7..3.1.213....3315.1.1.........
7-15  1.....11..............2..1..............11...3.1.1.2112...1.21....1...12.............4..3..12.2.....111.2.1..11.....
      ==========================================================================================================================
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2-14  ........1..1.................1....1......1...1..11..1.....1.1..1......2............8.26..5..34...31.1..1...92..2..
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7-19  ..............1.....................1...2.22...2........2.....22.2.2..1.1..............4.32..62.15...22.1......83..5..
2-21  .......1.........................1.....1..1.1.1..1...1.1.....11.1.....................8.21..41.47...2..2......631.3..
7-21  .......2.....12.......1.1........1.....1.11...2..1.........1.22.2.2..................1.8.12..42.27...21.2......62..4..
2-20  ....11.1..............11.........1.....1....1.1............2.16.2.4..................3..2.....17...41.3...1..52..3..
7-20  ....1..1....1.2.......11........1......1.2....2............2.1712.6..................3.14...11.8...31.3......52..3..
2-15  ......1...1..2.....1.............11.....11...........11.....11.1......1............1.7..2.13..22...1111.1....22..1..
```

4.8. Community ordination

4.8.1. Invertebrate Ordination

Only 9.9 percent of the variance could be explained by the first DCA axis, and a further 6.1 percent and 4 percent could be explained from the derivation of axis 2 and 3 (Table 4.8.1).

Table 4.8.1. Cumulative percentage of variance explained by the first few ordination axes.(Brackets show the correlation between variables).

Cumulative variance explained in...	Axis 1	Axis 2	Axis 3
Invertebrate ordination	9.9	16.0	20.0
Plant ordination	17.5	25.2	30.9
Invertebrate + environmental variables	8.4 (16.9)	12.9 (26.4)	16.1
Invertebrate + plant species data	8.5 (38.5)	12.8 (60.7)	15.6

Figure 4.8.1a shows site ordination axis 1 vs axis 2; (the sites are grouped according to the initial sampling regime). Site abbreviations are in Table 4.1.1a in the Species richness section. 'Shingle' sites appear at the top left corner of the ordination plot; 'Willow' sites at the bottom left, 'bulldozed' plus Recovering Treatment sites are across to the right of the plot along with Ohau site 3. The Ohau site 2 appears near the shingle sites; site 1 to the top of the plot in the centre. Figure 4.8.1b depicts changes in the sites with time (arrows join sites from 1993 to 1994). The bulldozed and Recovering Treatment sites have the longest arrows across axis 1. These sites were translocated the farthest across the plot from left to the right. Site 17 had the longest arrow. Axis 1 therefore represents a disturbance gradient from undisturbed (left) to disturbed (right).

In contrast, the majority of the Control sites (including the Ohau sites 1 and 2) showed very little movement on axis 1 but some change vertically on axis 2. Axis 2 probably has some seasonal influence (from flood-delayed sampling) as well as a gradient from open stable shingle (top) to heavily vegetated silty sites (bottom).

Figure 4.8.1c shows the sites grouped according to the invertebrate classification.

The species ordination diagram is shown in Figure 4.8.1d. (Species Identification codes are in Appendix 4). Although there is no clear clumping of invertebrate taxa into habitats, the stable-shingle taxa are more than four units of half change distant from species found in the Willow sites and bulldozed/disturbed sites. The cursorial Tigerbeetle (*Neocicindela feredayi*, species ID # 66) occurs across to the right with other organisms prominently found in Recovering Treatment and bulldozed habitats; Carabidae such as *Notagonum feredayi* (95) and *Bembidion* sp 1 (64) occur with invertebrates found in site 3 of the Ohau. Invertebrates in pasture-like (willow) areas occur towards the bottom left (e.g. *Costelytra zealandica* grass grub (41), and slug (219) (in the moister sites) while the species found primarily in stable Shingle sites such as the new species of *Hemiandrus* ground weta (25), the grasshopper *Sigaus minutus* (24), false scorpion (194) and ants (*Monomorium* spp, 11 and 205) occur towards the top left corner. Ubiquitous species blur habitat distinctions and the lack of distinctive 'habitat patterning' echoes the invertebrate classification two-way table. These species such as the introduced harvestman (*Phalangium opilio*, 21), native pest *Nysius huttoni* (6), mites (Acari, 1) and Collembola (26, 27, 28) occur in the centre of the ordination.

The positions of the Willow and Shingle sites are swapped on the third axis (Figure 4.8.1e) although the Ohau sites 1 and 2 maintain their positions high on the plot. Site 3 of the Ohau is displaced the most across this axis. Most of the bulldozed and Recovering Treatment sites do not show much change on axis 3. This axis emphasises the seasonal changes in the Control sites. Again, the Control sites are relatively static compared to the Experimental sites. The species ordination diagram is included in Appendix 7.

4.8.2. Plant Ordination.

17.5 % of the variance between sites was explained by the first DCA axis plant species, and in total the first three axes explained more than 30 percent (Table 4.8.1).

Figure 4.8.2a shows site ordination axis 1 vs axis 2; Site 1 of the Ohau is not shown as no plants were recorded from it at either date. Shingle sites appear at the right of the ordination plot, while Willow sites are uppermost in the centre; bulldozed sites show translocation towards the Recovering Treatment habitat which occurs in a central position below Willow sites. The Experimental Willow habitat before clearance is intermediate between the Recovering Treatment sites and the Control Willow sites. The Ohau site 2 lies between the Control Shingle sites and the Control Willow sites while the Ohau site 3 occurs across to the left.

Figure 4.8.2b shows the arrows of change in site position between sample dates. The Control Shingle and Control Willow sites do not shift far from their initial positions on axis 1 but have small amounts of vertical movement on axis 2; bulldozed and Recovering Treatment sites have longer arrows (similar to the invertebrate ordinations). Groupings of sites according to the plant classification are more concise than in the diagram of sites grouped according to the invertebrate ordination (Figure 4.8.2c). There are approximately four units of half change difference between sites on the extremes of axis 1 (which is almost twice the separation of sites according to the invertebrate data).

The axes gradients are more easily determined from the species ordination (Figure 4.8.2d). Axis 1 is probably a moisture and vegetation cover gradient; open dry sites with rock lichens, *Raoulia, Muehlenbeckia axillaris*, sheeps sorrel (*Rumex acetosella*), *Hieracium* spp, hares-foot treefoil (*Trifolium arvense*) and bryophytes appear on the right; lush, wet and pasture species such as the creeping buttercup (*Ranunculus repens*), *Mimulus guttatus* and *M. moschatus*, *Juncus articulatus*, sweet vernal (*Anthoxanthum odoratum*), *Prunella vulgaris* and willow appear to the left. Axis 2 is likely to be a substrate-disturbance related gradient; species such as *Epilobium melanocaulon*, *Raoulia tenuicaulis*, *Elymus rectisetus* which are characteristic native pioneer species on braided riverbeds occur at the upper end of axis 2; flowering weeds appearing on recently disturbed silty areas are closer to axis 1 (e.g. Californian poppy (*Eschscholzia californica*) and *Melilotus officinalis*).

Axis 3 emphasised the difference between the Ohau site 2 and the remaining sites (Figure 4.8.2e); the distinction between willow sites and those which have been bulldozed is less distinct while shingle sites are still recognisably separate. (This echoes the plant classification dendrogram). The plant species ordination for axis 1 vs axis 3 is shown in Appendix 7.

4.8.3. Invertebrate Ordination with Environmental variables

When environmental data was included as a secondary matrix, the amount of variance explained by the first two axes was similar to that of the sites when grouped by invertebrate data alone (Table 4.8.1). The grouping of sites was similar to the previous ordinations and the distinction between shingle, willow and bulldozed/Recovering Treatment area sites was more pronounced than by the invertebrate ordination alone (Figure 4.8.3a). Similar to the previous ordinations, the bulldozed and Recovering Treatment areas showed the most displacement along axis 1 while the Controls remained relatively static on all three axes.

The Environmental variable-invertebrate biplot for sites is shown in Figure 4.8.3b. The variables of willow proximity, shade, % cover of litter, bryophytes, vascular plants and bare ground had a large influence (long arrows) on distinguishing between sites. (Silt and sand had negligible variance and were omitted by the programme). The species ordination is shown in Appendix 7.

4.8.4. Invertebrate Ordination with Plant species data

This ordination also explained a similar amount of variance as the previous ordinations (Table 4.8.1) but the correlation between invertebrate taxa and plant variables was much higher than the correlation for invertebrate data and environmental variables.

The position of the control sites was relatively static compared to the bulldozed sites (Figure 4.8.4a). The Experimental Willow (pre-treatment) sites did not coincide with the Control Willow and (similar to the plant ordination alone) were intermediate in position between the Control Willow and the Recovering Treatment

sites. Similar to the invertebrate-environmental variable ordination, the Shingle habitats were distinct from the Willow habitats which were, in turn, distinct from the bulldozed and Recovering Treatment habitats. Site 1 and 3 of the Ohau occurred in close proximity to the bulldozed and Recovering Treatment habitats while site 2 occurred alongside the stable shingle sites.

The species diagram for this ordination is shown in Appendix 7. The moth species 144 was not located on the plot due to a problem in the programme. Because there were no plant species recorded from Ohau site 1, the program labelled the 1993 and 1994 sample of site 1 as 'passive'. Therefore, the secondary matrix no longer had similar dimensions to the primary matrix (34 sites instead of 36) which meant that invertebrate species which only occurred in site 1 were not registered by the programme. No solution could be found to this problem (Dr Ashley Sparrow, pers. comm).

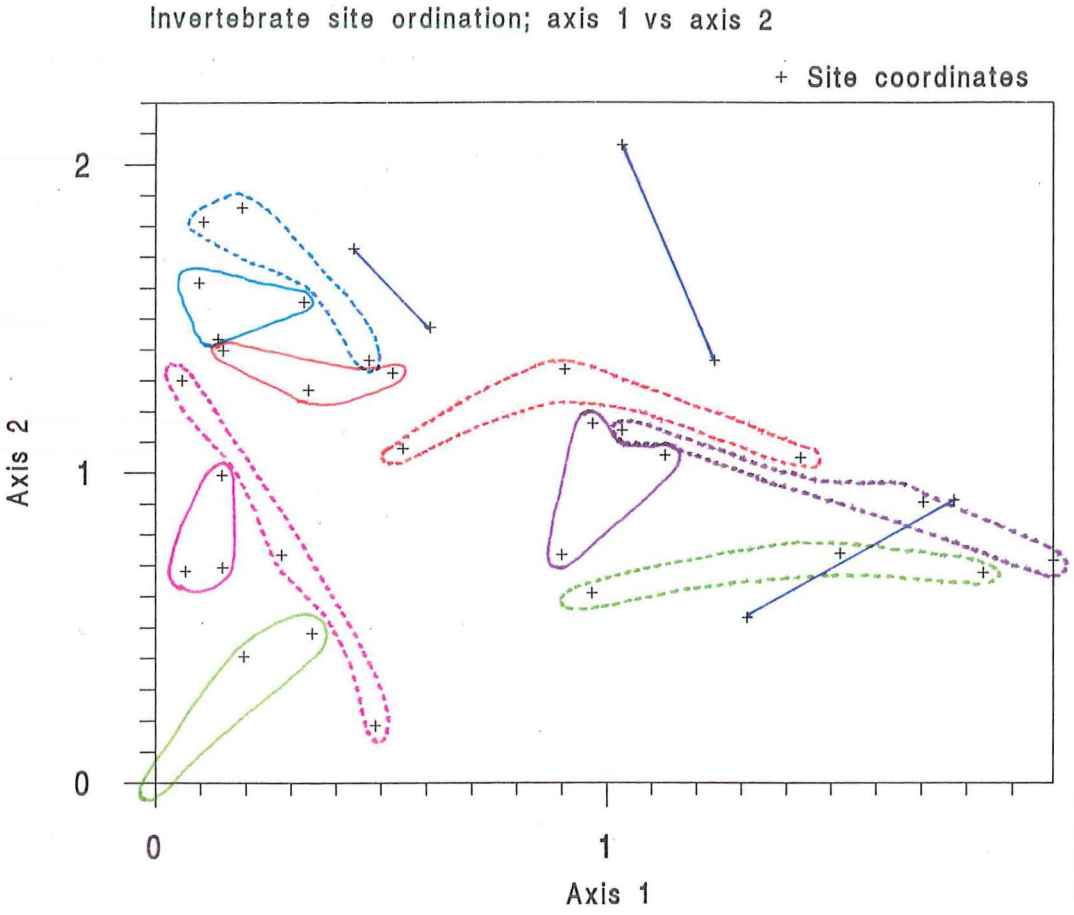

Figure 4.8.1a. Invertebrate site ordination, axis 1 vs axis 2. Sites are grouped according to th initial theoretical site groups. Solid lines = sites in the 1993 sample, broken lines = site in the 1994 sample. Axes are in units of half change.

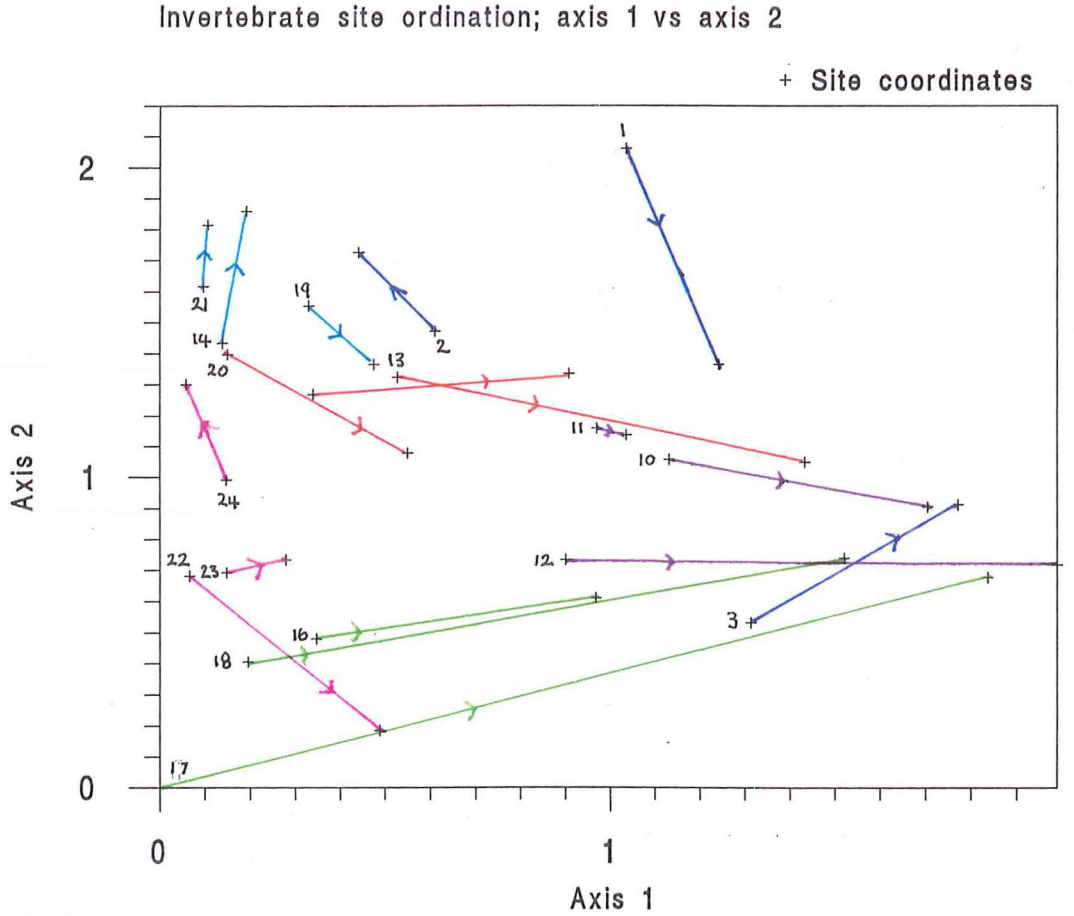

Figure 4.8.1b. Invertebrate site ordination axis 1 vs axis 2. Arrows connect sites from 1993 to 1994. The numbers at the base of each arrow are the site numbers. (See Table 4.1.1a for site labels). Axes are in units of half change.

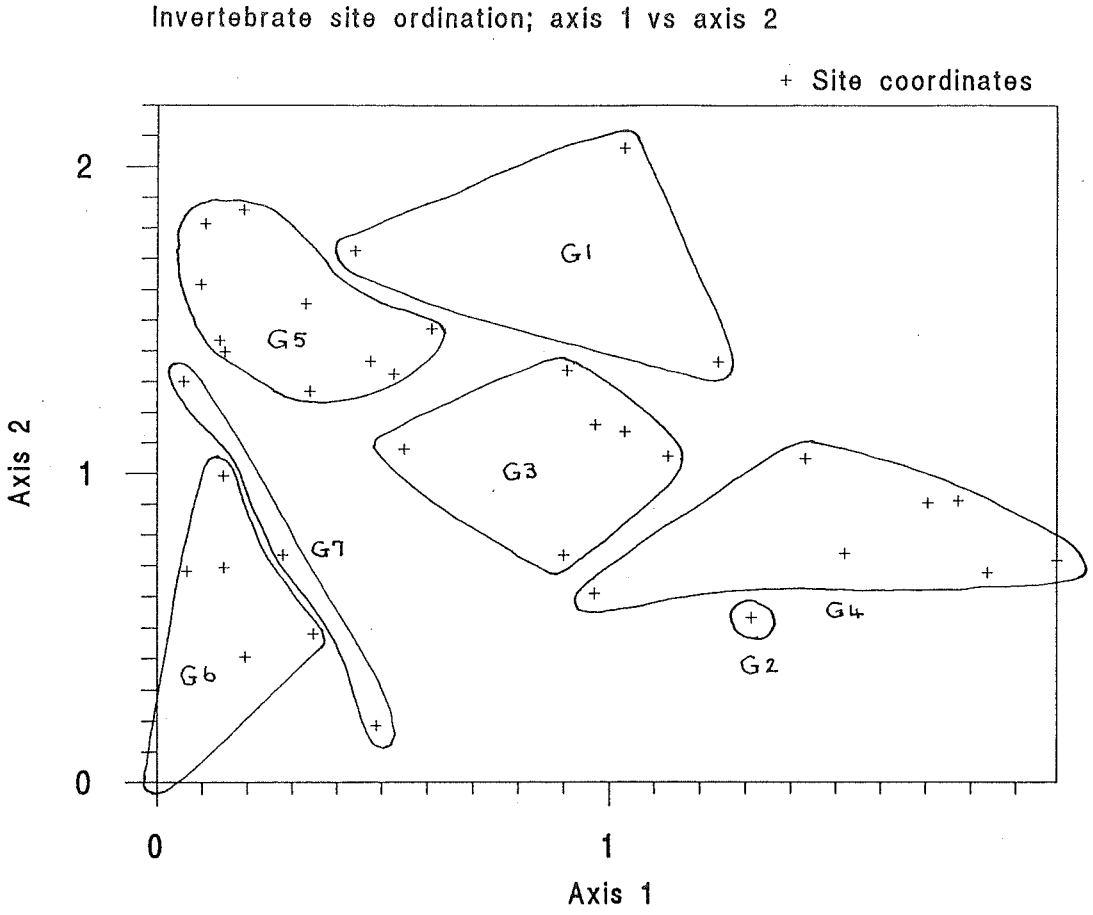

Figure 4.8.1c. Invertebrate site ordination, axis 1 vs axis 2. Sites are grouped according to the invertebrate classification. Axes are in units of half change.

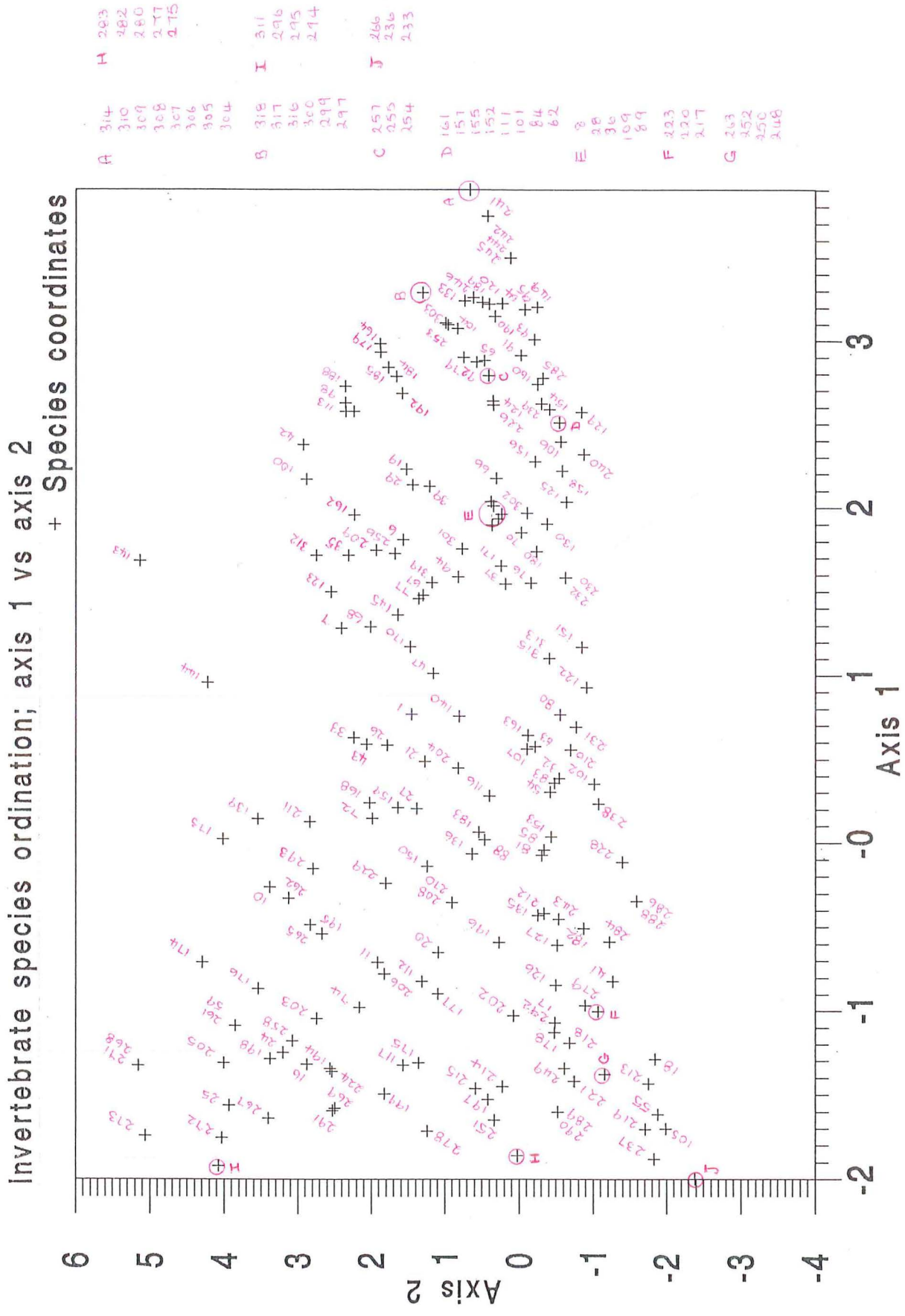

Figure 4.8.1d. Invertebrate species ordination, axis 1 vs axis 2. Numbers are species identification codes (these are in appendices). Axes are in units of half change.

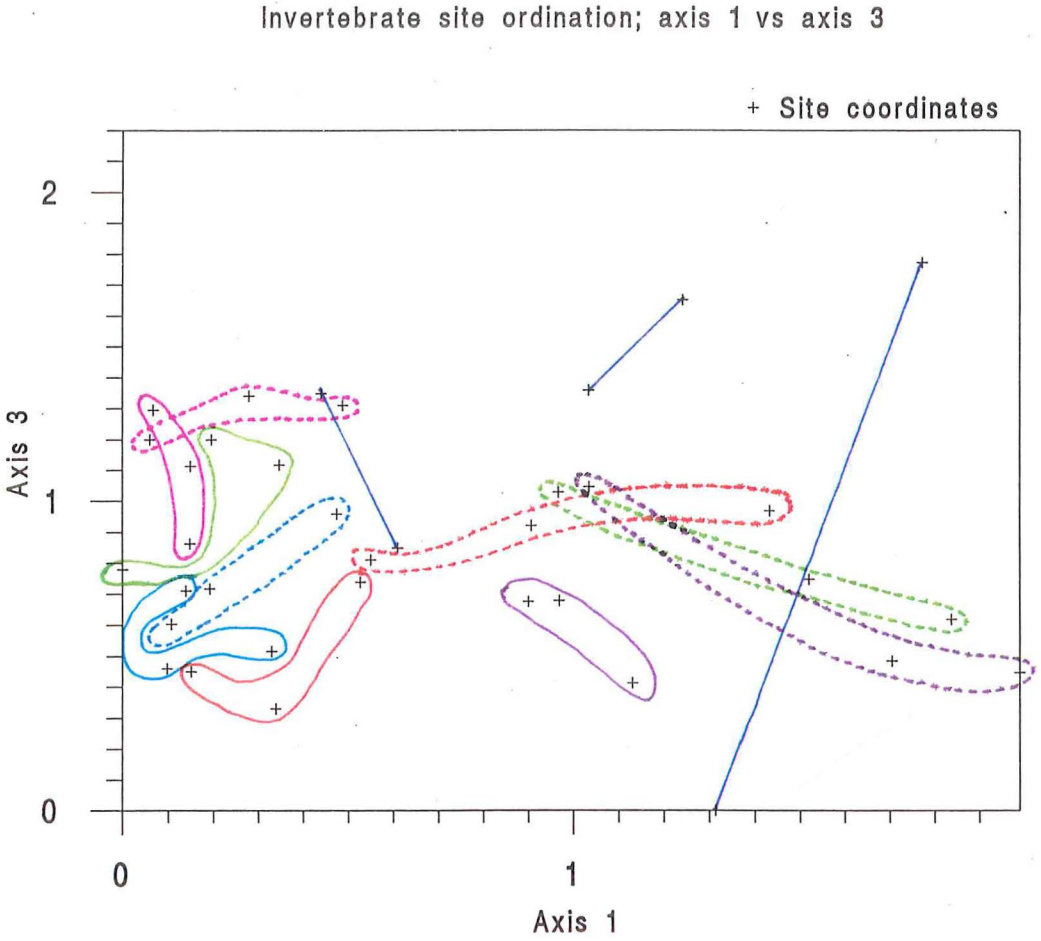

Figure 4.8.1e. Invertebrate site ordination, axis 1 vs axis 3. Sites are grouped according to the initial theoretical site groups. Solid lines = sites in the 1993 sample, broken lines = sites in the 1994 sample. Axes are in units of half change.

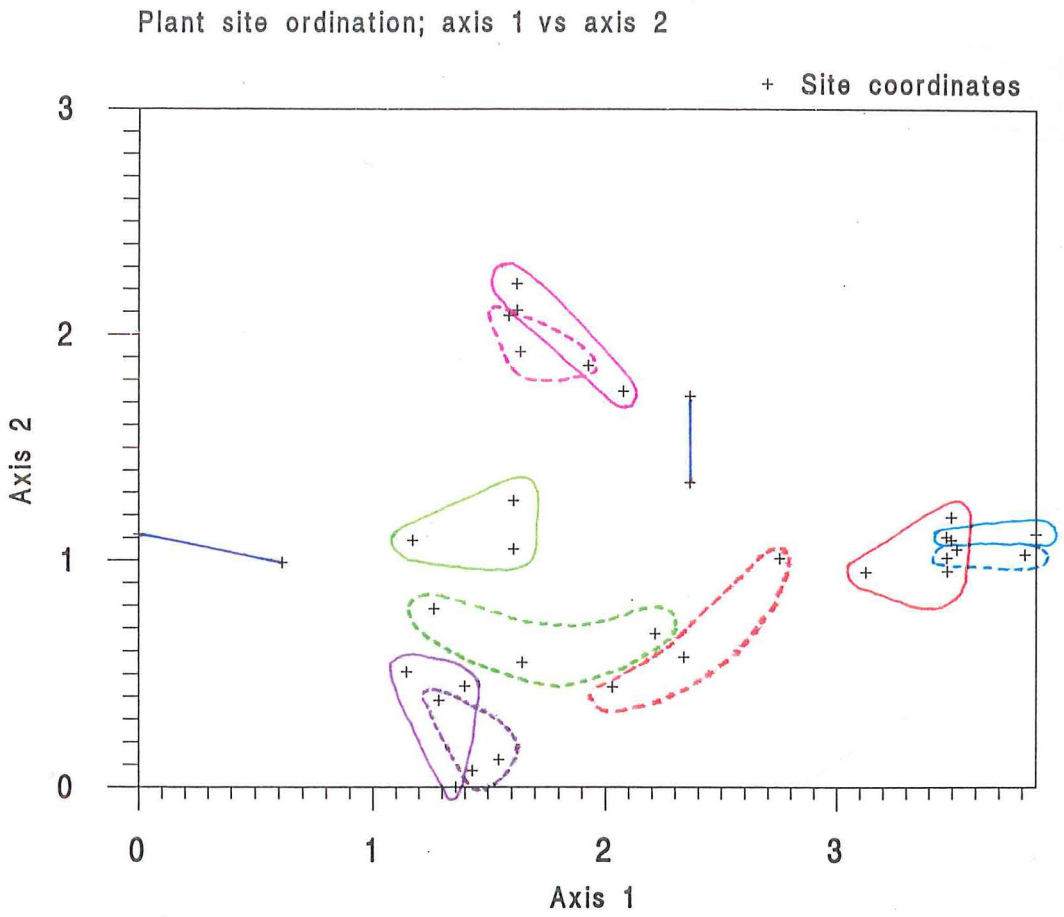

Figure 4.8.2a. Plant site ordination, axis 1 vs axis 2. Sites are grouped according to the initial theoretical site groups. Solid lines = sites in the 1993 sample, broken lines = sites in the 1994 sample. Axes are in units of half change.

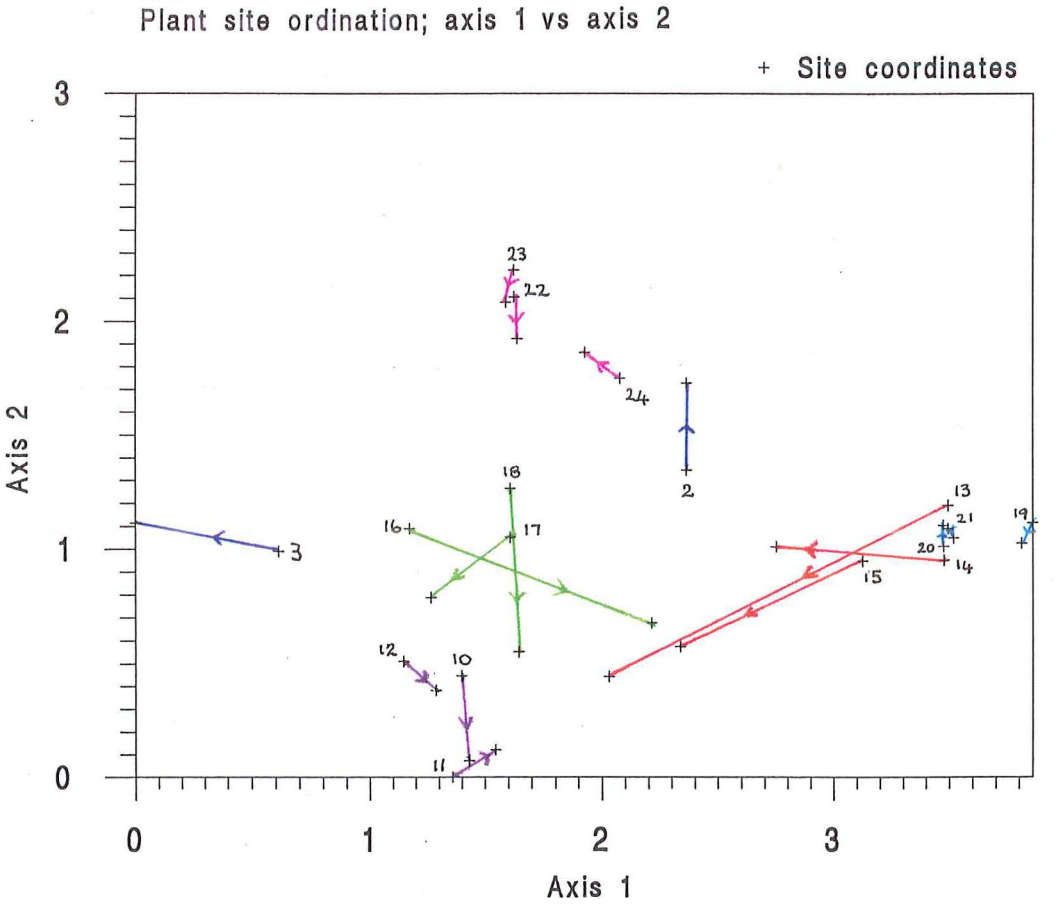

Figure 4.8.2b. Plant site ordination axis 1 vs axis 2. Arrows connect sites from 1993 to 1994. The numbers at the base of each arrow are the site numbers. (See Table 4.1.1a for site labels). Axes are in units of half change.

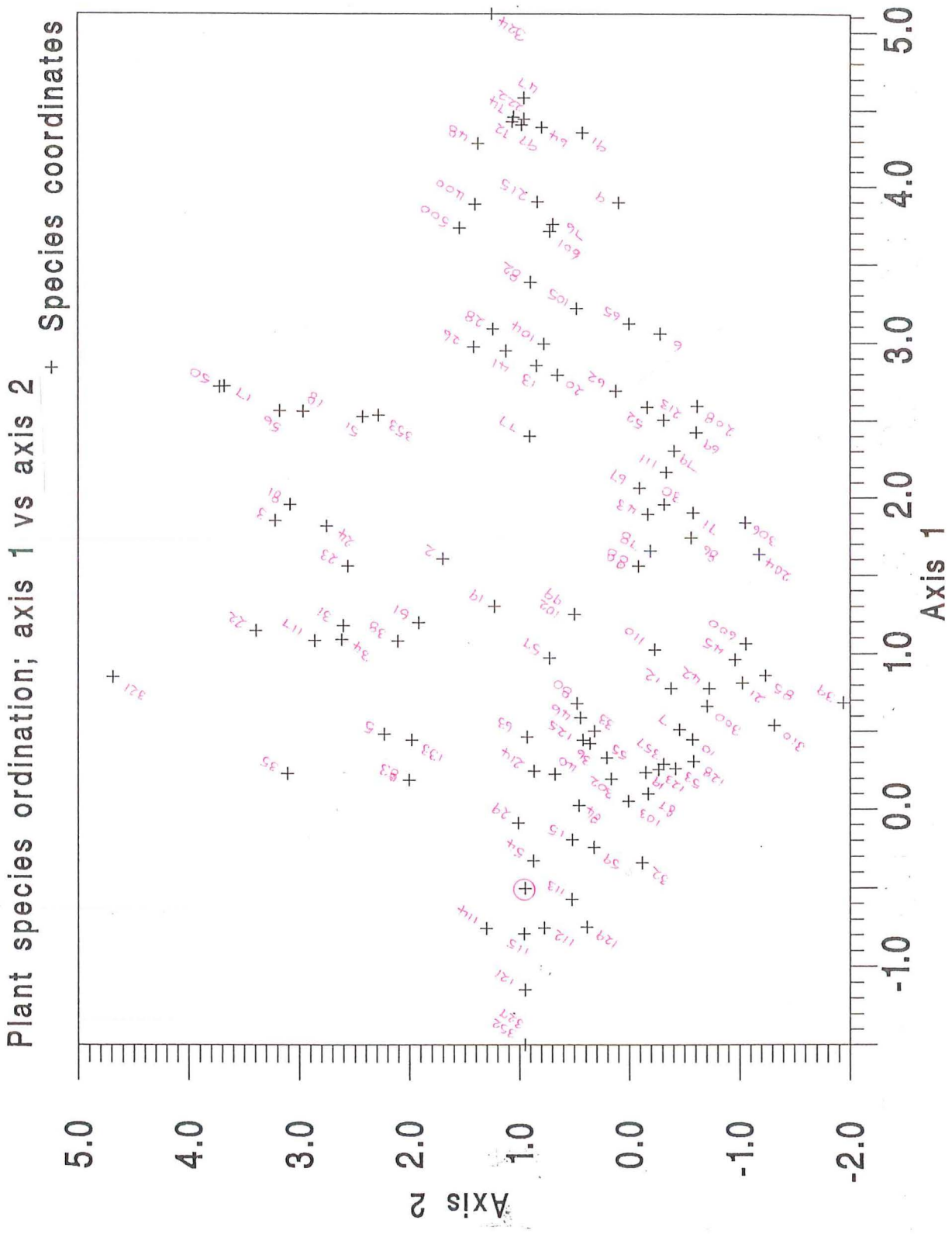

Figure 4.8.2d. Plant species ordination, axis 1 vs axis 2. Numbers are species identification codes (these are in appendices). Axes are in units of half change.

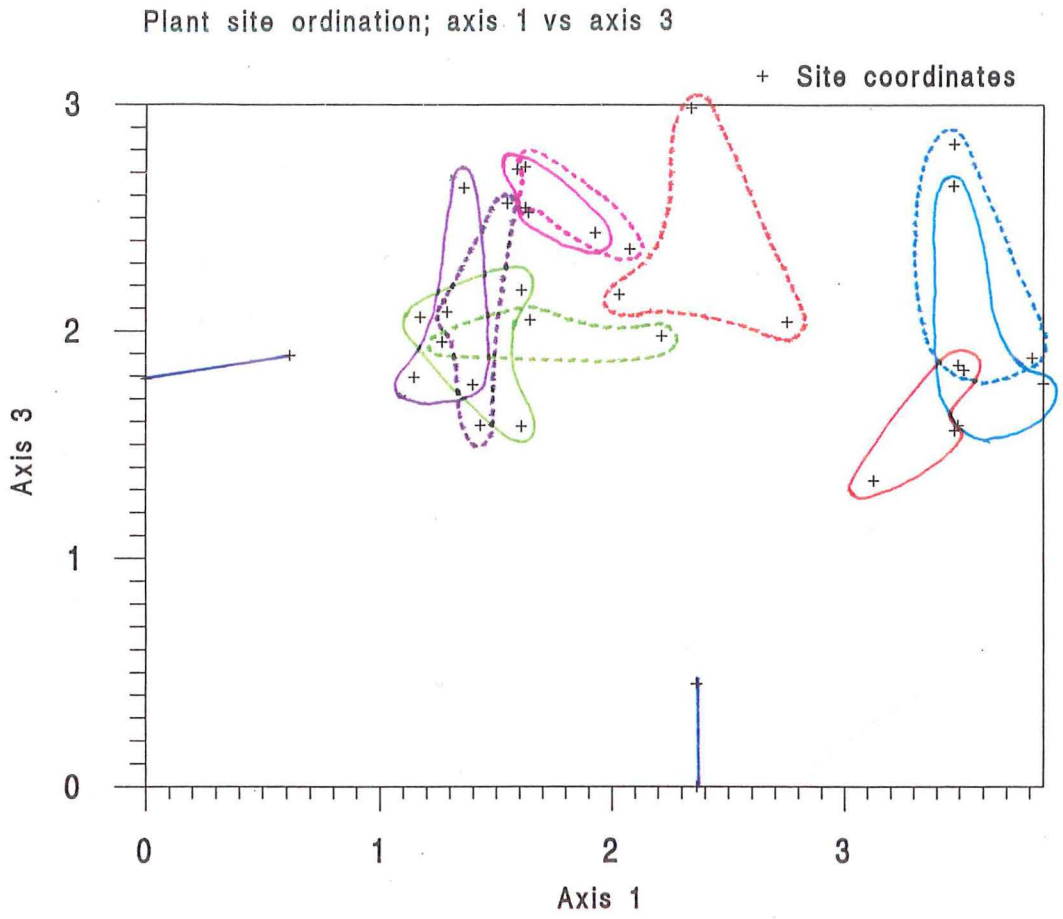

Figure 4.8.2e. Plant site ordination, axis 1 vs axis 3. Sites are grouped according to the initial theoretical site groups. Solid lines = sites in the 1993 sample, broken lines = sites in the 1994 sample. Axes are in units of half change.

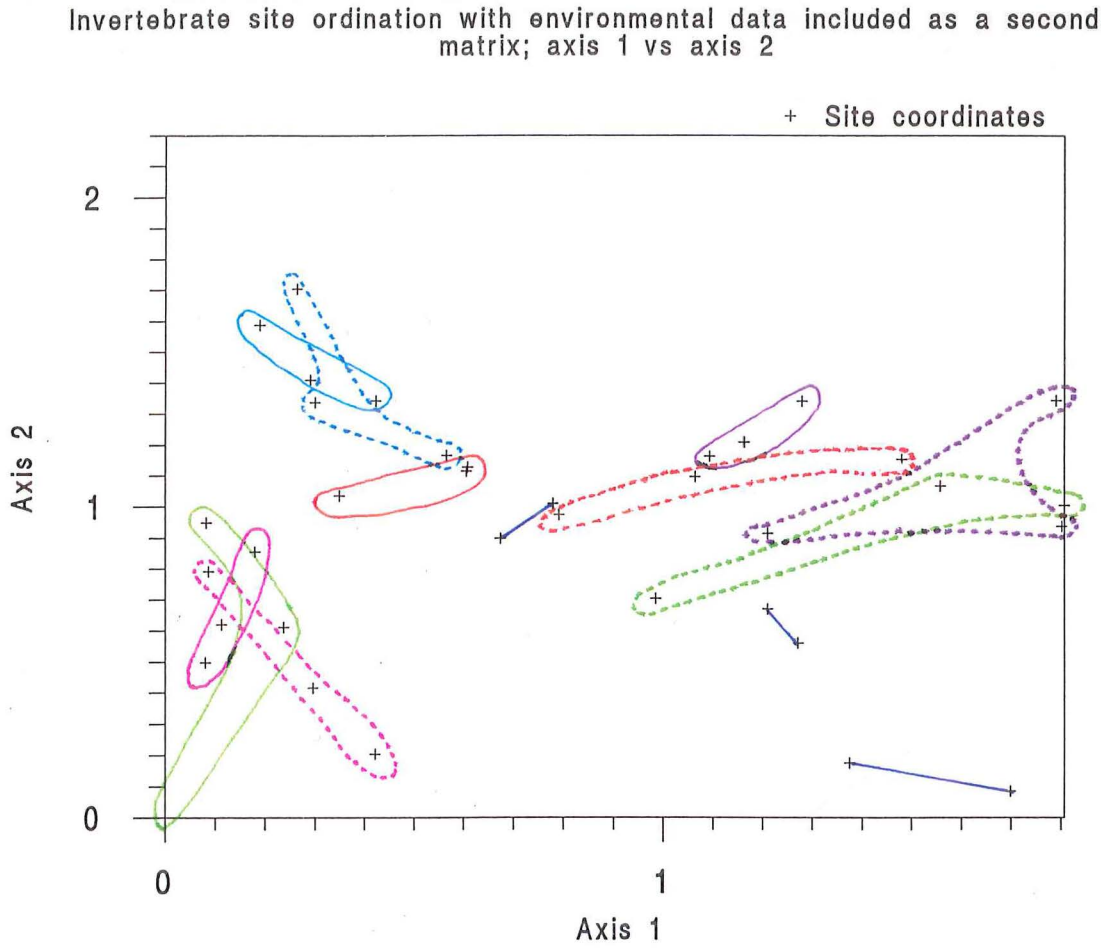

Figure 4.8.3a. Invertebrate site ordination with environmental variables included as a secondary matrix, axis 1 vs axis 2. Sites are grouped according to the initial (theoretical) groups. Axes are in units of half change.

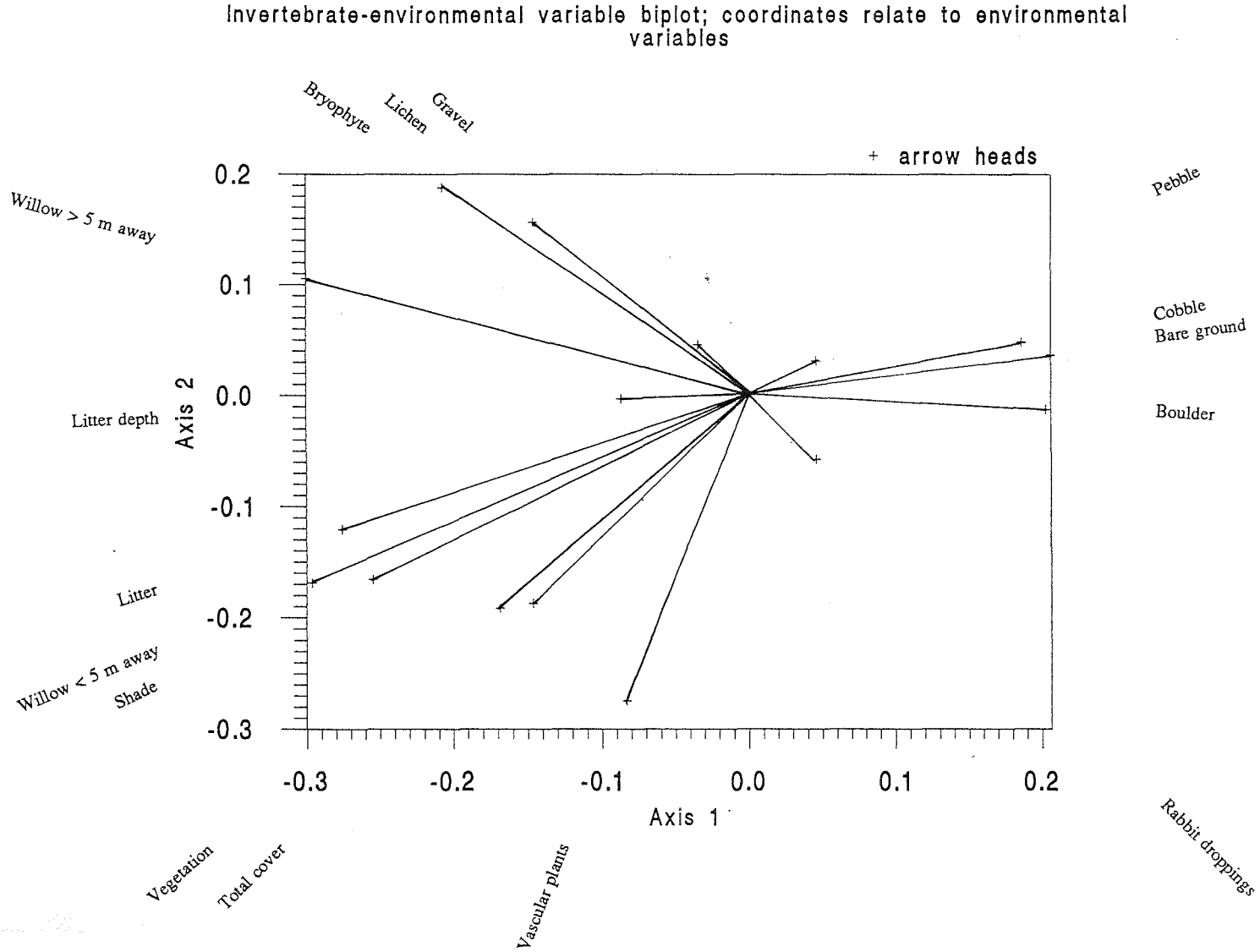

Figure 4.8.3b. Invertebrate-environmental variable biplot. Variables with the longest arrows explain most of the variation in the invertebrate data set.

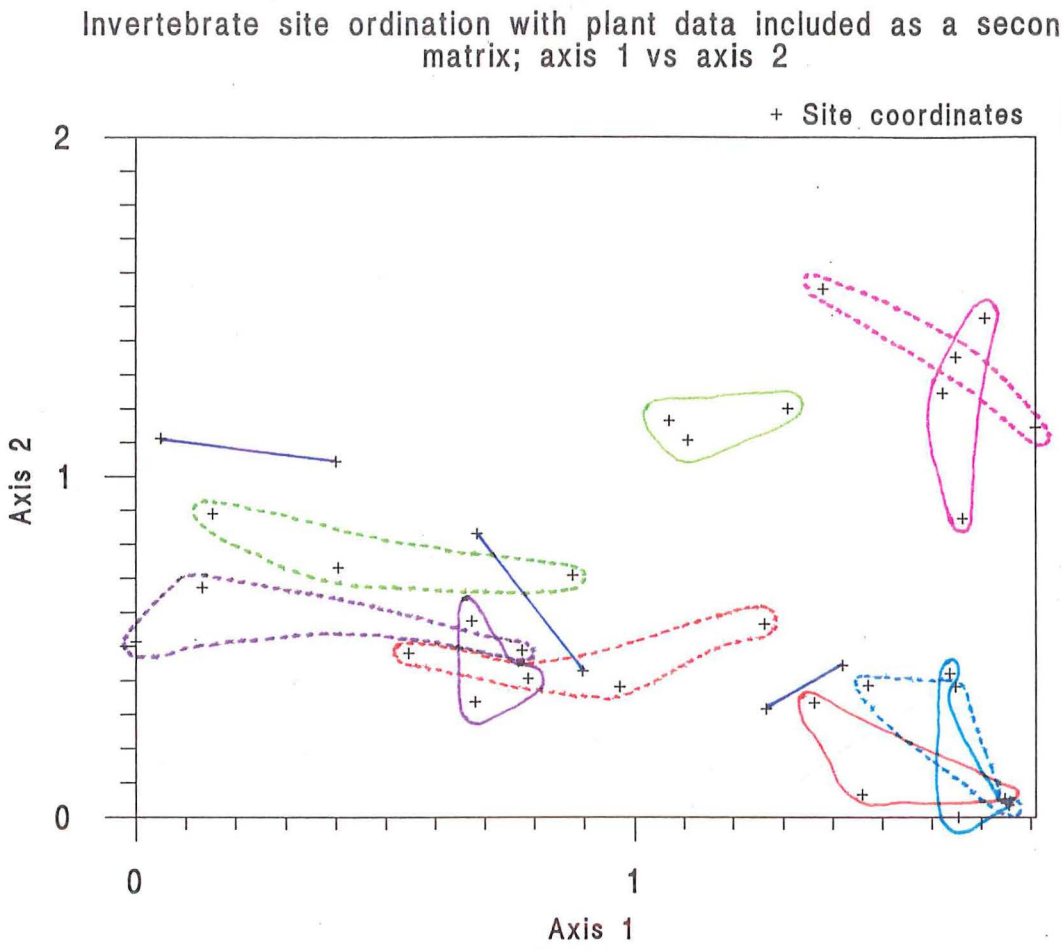

Figure 4.8.4. Invertebrate ordination with plant species included as a secondary matrix, axis 1 vs axis 2. Sites are grouped according to the initial theoretical site groups. Axes are in units of half change.

INVESTIGATION OF SPECIES DATA

4.9. Plants and Invertebrates as indicators of improvement in habitat quality; change towards the Ohau 'braided' state

4.9.1. Invertebrates

Before bulldozing, the sample from the **Experimental Shingle** habitat contained 53 taxa (Table 4.9.1). 32 of these taxa were also represented in the Ohau which could already indicate braided habitat. However, all 32 taxa were found in other control habitats (including the Experimental Willow habitat before clearance) and were therefore not exclusive to a braided habitat (Ohau) plus the Experimental Shingle habitat before clearance. Therefore, there were no specific braided-type taxa in the Experimental Shingle habitat before clearance.

After bulldozing, the sample from the Experimental Shingle habitat contained 84 taxa (Table 4.9.1). 52 were also represented in the Ohau habitat (at either sample date) which could potentially indicate a braided-type habitat. However, 29 of these 52 taxa appeared in the Experimental Shingle sample BEFORE bulldozing as well as after. These were considered resilient (unaffected by bulldozing). Two further species occurred only in the Control habitats in the 1994 sample. These represented seasonal species (detected in the flood-delayed sample). Of the remaining 21 species, 13 were also found in the Control sites and were therefore not exclusive to the Ohau (braided) habitat plus the Treatment area. The remaining 10 taxa were exclusive to the Experimental Shingle area (post-treatment) and the Ohau habitat. These were considered 'braided riverbed taxa' which appeared in the Experimental Shingle habitat after clearance. They are listed in Table 4.9.1a. (Taxa which occurred in site 3 of the Ohau habitat are considered as prey which is accessible to Wrybills).

The sample from the **Experimental Willow** habitat before bulldozing contained 87 taxa (Table 4.9.1). 41 of these were also found in the Ohau habitat. However, 36 of these 41 taxa were found elsewhere (in control habitats and Experimental Shingle before clearance). Five species remained which occurred in the Experimental Willow area before clearance plus the braided Ohau habitat. These

represented 'braided riverbed taxa' already present in the Experimental habitat before willow clearance. These are listed in Table 4.9.1b.

After bulldozing, the sample from the Experimental Willow habitat contained 93 taxa, 53 of which were also represented in the Ohau habitat. 36 occurred in the Experimental Willow area BEFORE clearance and were considered resilient, and a further two were seasonal. Of the remaining 15 species, five were found in other habitats. The ten remaining taxa were therefore exclusive to the Ohau (braided) habitat plus the Experimental Willow habitat post-clearance. These are listed in Table 4.9.1c. (Taxa which occurred in site 3 of the Ohau habitat are considered as prey which is accessible to Wrybills).

The 1993 sample of the **Recovering Treatment area** contained 63 taxa (Table 4.9.1). 41 species were also represented in the Ohau. However, 37 of these taxa were found in the Control sites (including the Experimental areas before clearance) and were considered ubiquitous (some examples of these ubiquitous taxa are listed in Table 4.4.3e). The four remaining species were exclusive to the Recovering Treatment habitat plus the Ohau habitat one year after willow clearance. These taxa are listed in Table 4.9.3d.

In 1994, the sample from the Recovering Treatment area contained 84 species, 49 of which were also represented in the Ohau. Eight appeared in the Control habitats only in the 1994 sample and their appearance was considered seasonal. 31 taxa were also found in other Control areas plus the pre-bulldozed sites. Ten species remained which were exclusive to the Ohau (braided) habitat plus the Recovering Treatment area in the second year following willow clearance. These species are also listed in Table 4.9.1d. (Taxa which occurred in site 3 of the Ohau habitat are considered as prey which is accessible to Wrybills).

4.9.2. Plants

The sample from the **Experimental Shingle** habitat before willow clearance contained 48 plant taxa (Table 4.9.2). 32 were also represented in the Ohau habitat. However, 31 of these common taxa were found in other control sites and therefore

were not exclusive to the braided Ohau plus the Experimental Shingle habitat. One species was already exclusive to the Ohau and the Experimental habitat before bulldozing (Table 4.9.2a).

After clearance, the sample from the Experimental Shingle habitat contained 57 plant taxa. 35 of these were also represented in the Ohau habitat. However, 25 of the 35 taxa were present in the treatment area BEFORE clearance and were considered resilient. One species appeared in the Control sites in 1994 and its presence was considered seasonal and five were recorded from other control areas. Four taxa remained which were exclusive to the Experimental Shingle habitat before clearance plus the Ohau habitat. These represented braided riverbed species which arrived in the Experimental Shingle habitat after clearance. They are listed in Table 4.9.2b.

The sample from the **Experimental Willow** habitat before clearance contained 59 plant taxa, of which 42 were also represented in the Ohau habitat (Table 4.9.2). 35 plant taxa were found in other control habitats so their appearance was therefore not restricted to a braided habitat. The seven remaining plant taxa that were exclusive to the braided Ohau habitat plus the Experimental Willow habitat before clearance are listed in Table 4.9.2b.

After willow clearance, the sample from the Experimental Willow habitat contained 61 plant taxa and 39 of these were also found in the Ohau habitat. Of these, 31 were found in this habitat before clearance and a further three species appeared in the Control habitats in 1994 sample. These were resilient and seasonal taxa respectively. Two were found in other control sites. Three species appeared in the Experimental Willow habitat AFTER clearance plus the braided Ohau. These are listed in Table 4.9.2c.

The 1993 sample of the **Recovering Treatment area** contained 58 species, 40 of which were also represented in the Ohau (Table 4.9.2). 35 taxa were also found in other control habitats (some examples of these ubiquitous species are listed in Table 4.9.2e). The five remaining species represented 'braided riverbed taxa' which

appeared in the Recovering Treatment area one year after willow clearance. These are listed in Table 4.9.2d. The eight 'braided riverbed taxa' which were found in the 1994 sample from the Recovering treatment habitat are also listed in this table.

Table 4.9.1. Invertebrates as indicators of improvement in habitat quality (change towards the Ohau state), measured by the appearance of braided habitat species in the Experimental and Recovering Treatment habitats. (Empty cells denote categories are not applicable for that habitat).

	Expt Shingle 1993	Expt Shingle 1994	Expt Willow 1993	Expt Willow 1994	Rec Trt 1993	Rec Trt 1994
Number of taxa found in habitat	53	84	87	93	63	84
Number of taxa shared with Ohau habitat	32	52	41	53	41	49
Number of taxa found before as well as after clearance (resilient species)		29		36		
Number of taxa found in Control habitats in 1994 only (seasonal species)		2		2		8
Number of remaining shared taxa		21		15		41
Number of taxa also found in Control habitats (both dates)	32	11	36	5	37	31
Number of species exclusive to this habitat plus the Ohau habitat ('braided riverbed taxa')	0	10	5	10	4	10

Table 4.9.1a. *'Braided riverbed'* invertebrate taxa represented in the Experimental Shingle habitat after bulldozing (1994). Numbers below the habitat headings depict the abundance of each taxa. The column headed 'site 3 only' are those taxa (*) found in the Ohau habitat in site 3 only and could represent taxa available to Wrybills as food.

Species Id #	Taxa identification	Expt Shingle 1994	Ohau 1993	Ohau 1994	Site 3 of Ohau	Trophic status
	COLEOPTERA					
42	*Actenonyx bembidioides*	2	91	30		predator
64	*Bembidion* sp 1	1	4	2	*	predator
124	*Coccinella undecimpunctata*	3		2	*	predator
125	Byrrhidae sp 2	1	9		*	lichen/fungi
123	Staphylinidae sp 3	1	2			predator
	HYMENOPTERA					
159	Pteromalidae sp 1	1	1		*	parasitic
179	Bethylidae sp 1	1		1	*	parasitic
	ARANEAE					
100	Agelenidae sp 1	1	2			predator
139	Hahniidae sp 1	1	3	7		predator
	LEPIDOPTERA					
39	Noctuidae	1	1	1		herbivore

Table 4.9.1b. *'Braided riverbed' invertebrate taxa represented in the Experimental Willow habitat before bulldozing (1993). Numbers below the habitat headings depict the abundance of each taxa. The taxa also found in the Experimental Willow habitat after bulldozing represent resilient taxa (unaffected by bulldozing).*

Species Id #	Taxa identification	Expt Willow 1993	Ohau 1993	Ohau 1994	Found in Expt W 1994 also
	COLEOPTERA				
125	Byrrhidae sp 2	4	9		*
	ARANEAE				
151	Dictynidae	1	1		
	HEMIPTERA				
153	Cercopidae sp 1	2	1	1	
158	Cicadellidae sp 2	3	1	2	*
	ANNELIDA				
129	Earthworm	1	1		

Table 4.9.1c. 'Braided riverbed' invertebrate taxa represented in the Experimental Willow habitat after bulldozing (1994). Numbers below the habitat headings depict the abundance of each taxa. The column headed 'site 3 only' are those taxa (*) found in the Ohau habitat in site 3 only and could represent taxa available to Wrybills as food.

Species Id #	Taxa identification	Expt Willow 1994	Ohau 1993	Ohau 1994	Site 3 only	Trophic status
	COLEOPTERA					
42	*Actenonyx bembidioides*	7	91	30		predator
64	*Bembidion* sp 1	9	4	2	*	predator
95	*Notogonum feredayi*	6	8		*	predator
154	*Hypharpax* sp 1	1	5		*	predator
93	Byrrhidae sp 1	2	11		*	lichen/fungi
124	*Coccinella undecimpunctata*	2		2	*	predator
120	Colydiidae sp 1	1	2	21		? fungi
	HYMENOPTERA					
164	*Stethynium* sp	1		2	*	parasite
253	*Trichopria* sp 4	1		2	*	parasite
	NEUROPTERA					
226	Neuroptera	2		16	*	predator

Table 4.9.1d. 'Braided riverbed taxa' in the Recovering Treatment area one and two years after willow clearance. Numbers below the habitat column headings are the abundance of each taxa. The column headed 'site 3 only' are those taxa (*) found in the Ohau habitat in site 3 only and could represent taxa available to Wrybills as food.

Species Id #	Taxa identification	Rec Trt 1993	Rec Trt 1994	Ohau 1993	Ohau 1994	Site 3 only	Trophic status
	COLEOPTERA						
42	*Actenonyx bembidioides*	2	2	91	30		predator
64	*Bembidion* sp 1	2	33	4	2	*	predator
95	*Notogonum feredayi*		8	8		*	predator
93	Byrrhidae sp 1		3	11		*	lichen/fungi
120	Colydiidae sp 1		2	2	21	*	lichen/fungi
124	*Coccinella undecimpunctata*	4	3		2	*	predator
	ARANEAE						
100	Agelenidae sp 1		1	2			predator
	DIPTERA						
149	*Nematopus* sp		1	1		*	nectivore
113	Muscidae sp 1		2		4		bacteria feeder
	NEUROPTERA						
226	Neuroptera		2		16	*	predator
	LEPIDOPTERA						
39	Noctuidae	2		1	1		herbivore

Table 4.9.1e. Some examples of ubiquitous invertebrate taxa present in the majority of habitats. I = introduced, N = native.

Species Id #	Taxa identification	COMMENTS on taxa
	DIPTERA	
37	Sciariidae	N; lumped category, feeds on rotten veg
116	*Scatella* new sp	N (undescribed); winged and wingless morphs; the macropterous form appeared after clearance
135	Cecidomyidae	I + N; lumped category, feeds on rotten veg
	HYMENOPTERA	
11	*Monomorium antarcticum*	N; social (ant), fine particle ground
74	*Trichogramma* sp	N; parasitises spider eggs; is minute + brachypterous
150	*Priocnemis nitidiventris*	N; cursorial spider hunter; digs burrows for prey (parasitic)
	ARANEAE	
35	Toxopidae sp 1	cursorial; N
83	Dictynidae sp 2	cursorial; N
112	Theridiidae sp 1	small; N?
140	Gnaphosidae sp 3	N?
	HEMIPTERA	
6	Lygaeidae sp 1	N; pest on crops, feeds on rosette, seeds and apical stem
36	Aphididae sp 2	I; winged and wingless forms caught
88	Margarodidae	N
	MISCELLANEOUS	
1	Acari	N?; lumped category
26	Entomobryoidea Collembola	lumped category, feed on unicellular bacteria and fungi
27	Poduroidea Collembola	small, high densities; lumped category, ditto
28	Neelipleona/ Symphypleona Collembola	lumped category, ditto
21	*Phalangium opilio*	I; harvestman, predator, scavenger
63	Gryllidae	N; vegetarian
85	Lepidoptera larvae	lumped category
89	Coleoptera larvae	lumped category (majority were Carabidae), predator

94	Thysanoptera	I; lumped category; winged and wingless forms caught, pollen feeders
126	*Forficula auricularia*	I; scavenger

Table 4.9.2. Plants as indicators of an improvement in habitat quality (change towards the Ohau state), measured by the appearance of braided habitat species in the Experimental and Recovering Treatment habitats. (Empty cells denote categories are not applicable for that habitat).

	Expt Shingle 1993	Expt Shingle 1994	Expt Willow 1993	Expt Willow 1994	Rec Trt 1993	Rec Trt 1994
Number of taxa found in habitat	48	57	59	61	58	61
Number of taxa shared with Ohau habitat	32	35	42	39	40	46
Number of taxa found before as well as after clearance (resilient species)		25		31		
Number of taxa found in Control habitats in 1994 only (seasonal species)		1		3		
Number of remaining species		9		5		
Number of species also found in Control habitats (both dates)	31	5	35	2	35	38
Number of species exclusive to this habitat plus the Ohau habitat ('braided riverbed taxa')	1	4	7	3	5	8

Table 4.9.2a. 'Braided riverbed' plant taxon represented in the Experimental Shingle habitat before bulldozing (1993).

Species Id #	Taxa identification
32	*Juncus articulatus*

Table 4.9.2b. 'Braided riverbed' plant taxa represented in the Experimental Shingle habitat after bulldozing (1994).

Species Id #	Taxa identification
53	*Rumex* cf *crispus*
112	*Achillea mellefolium*
133	*Lotus corniculatus*
353	*Poa* 'blue' sp

Table 4.9.2c.'Braided riverbed' plant taxa represented in the Experimental Willow habitat before bulldozing (1993).

Species Id #	Taxa identification
19	*Epilobium* cf *ciliatum*
31	*Hypochoeris radicata*
33	*Juncus* cf *tenuis*
40	*Mimulus guttatus*
54	*Sagina procumbens*
84	*Cirsium arvense*
103	*Isolepis setacea*

Table 4.9.2d. 'Braided riverbed' plant taxa represented in the Experimental Willow habitat after bulldozing.

Species Id #	Taxa identification
53	*Rumex* cf *crispus*
115	*Ranunculus repens*
128	*Rumex* cf *obtusifolius*

Table 4.9.2e. 'Braided riverbed' plant taxa represented in the Recovering Treatment area two years after clearance.

Species Id #	Taxa identification
15	*Discaria toumatou*
53	*Rumex* cf *crispus*
59	*Stellaria graminea*
112	*Achillea millefolium*
113	*Trifolium pratense*
123	*Navarretia squarrosa*
128	*Rumex obtusifolius*
129	*Juncus effusus*
133	*Lotus corniculatus*
353	*Poa* 'blue' sp

Table 4.9.2f. Some examples of ubiquitous plant taxa recorded in the majority of habitats.
I = Introduced, N = native

Species Id #	Taxa Identification	COMMENTS on taxa
2	*Agrostis capillaris*	I; Browntop grass
26	*Hieracium* spp.	I; Hawkweed
28	*Hieracium pilosella*	I; Hawkweed
41	*Muehlenbeckia axillaris*	N; colonises bare shingle but also present in sward under willows
43	*Plantago lanceolata*	I; Plantain
51	*Rosa rubiginosa*	I; Sweet brier
62	*Trifolium arvense*	I; Hares-foot tree foil (legume)
65	*Verbascum thapsis*	I; Flannel leaf
77	*Echium vulgare*	I; Viper's bugloss
81	*Poa* spp	N?
500	Unid. Bryophytes	N?

4.10. Taxa which were lost from the Experimental habitats

4.10.1. Invertebrates

48 invertebrate taxa were not recorded from the 1994 sample of the bulldozed (Experimental) habitats (Table 4.10.1). However, 13 of these taxa were only represented in the Experimental habitats; 15 taxa were lost from other areas as well, and their loss represented a seasonal effect. Three taxa appeared in other sites but were lost from the Experimental areas which also represented a seasonal effect (Appendix 8). 17 species remained whose absence could be indicative of a negative impact of willow clearance. These are listed in Table 4.10.1a. The most number of taxa was lost from the order Hymenoptera (Table 4.10.1b).

4.10.2. Plants

24 plant taxa were lost from the Experimental habitats (Table 4.10.1). Six species were represented only in the Experimental sites (Appendix 8), and a further one species represented a seasonal loss. 17 species remained whose absence could be due to willow clearance (Table 4.10.2a).

4.11. Invertebrates as predator food

Twenty-two invertebrate taxa were eaten by the cats and ferrets in Amelia Pascoe's study (n = 25 ferrets, 53 cats and 56 cat scats). With her permission, these are listed in Table 4.11.1 (in descending numerical order). Thirteen of these taxa were caught in pitfall traps in the present study (marked with an asterisk.)

Three ferret guts contained invertebrates (one of which contained the majority of the *Dermestes* larvae). Invertebrates occurred in 53 % of cat guts and less than 20 % of the cat scats; they contributed < 1 % to their diet by weight. Some predators relied heavily on invertebrates as prey. The gut from a 1.3kg female cat contained only invertebrates; these were 14 *Hemideina maori* tree-weta, four large Noctuidae caterpillars, a *Hemiandrus* ground-weta, and a large *Hexathele* funnel-web spider.

Some invertebrates were not caught in this study as they were not susceptible to capture using a pitfall trap technique. For example, the *Dermestes* 'bacon beetle' larvae feed on dry carcasses and as the majority of this prey item was consumed by one ferret they may represent a bias in that animal's diet; the large *Hexathele* funnel web spider has been seen in the study area, but may have been able to avoid the traps; The *Prodontria* beetle was present in the spring and autumn samples, but not the summer samples analyzed for this project; the aquatic caddis fly larvae were possibly ingested while the predators were drinking.

The tree weta *Hemideina maori*, the burrowing ground weta *Hemiandrus* and the large *Hexathele* funnel web spider were eaten by more than one predator; these taxa probably represent a deliberate part of the predators' diet. Changes in abundance of the two weta species' found in this study are summarised in Table 4.11.2.

Table 4.11.2. Weta abundance per habitat. (Blank cells indicate absence).

Habitat	*Hemideina maori*		*Hemiandrus* new sp	
	1993	1994	1993	1994
Expt Sh	1	2	2	
Expt W	1			
Cont Sh	1		10	7
Cont W	1		1	

Table 4.10.1. Number of taxa lost from the Experimental habitats after bulldozing.

	Number of invertebrate taxa	Number of plant taxa
Total number of taxa lost from Experimental habitats after bulldozing	48	24
Number of taxa represented only in Experimental habitat	13	6
Seasonal loss of species	15	1
Seasonal gain of species	3	
Remaining species lost from Experimental habitats; (potential indicators of habitat disturbance)	17	17

Table 4.10.1a. Number of invertebrate taxa lost from the Experimental habitats after bulldozing. Miscellaneous refers to taxa from Isopoda, Mollusca, Annelida, Turbellaria, Arachnida (excluding spiders), Ephemeroptera, Myriopoda.

Order	Number of taxa
Hymenoptera	20
Diptera	7
Coleoptera	4
Hemiptera	3
Orthoptera	3
Araneae	2
Miscellaneous	8

Table 4.10.1b. Invertebrate taxa lost from Experimental habitats possibly as a result of bulldozing.

Species Id #	Taxa identification	COMMENTS about the organisms
	COLEOPTERA	
117	Oedomeridae	adults eat fungi and pollen
215	*Demetrida dieffenbachi*	predator
	HYMENOPTERA	
163	Braconidae spp	parasitic
171	*Trichopria* sp 3	parasitic
	DIPTERA	
199	Tachinidae sp 2	
107	*Smittia* sp	aquatic; aerial plankton?
249	*Neolimnia ?minuta*	
	HEMIPTERA	
72	Unid. juvenile Hemiptera	
153	Cercopidae sp 1	feeds on plant liquid
	ORTHOPTERA	
25	*Hemiandrus* new sp	burrowing, endemic
206	*Phaulacridium marginale*	native
	ARACHNIDA	
221	Stiphidiidae	
194	Pseudoscorpiones	predator
	ISOPODA	
175	*Porcellio scaber*	litter associated habitat, introduced
	MOLLUSCA	
127	Small snail	wet vegetation
219	Slug	wet vegetation
	ANNELIDA	
129	Earthworm	requires stable soil structure

Table 4.10.2. Plant taxa lost from Experimental habitats possibly as a result of bulldozing.

Species Id #	Taxa identification	COMMENTS on plants
22	*Festuca ovina*	confusion with *rubra*?
24	*Gonocarpus aggregatus*	
31	*Hypochoeris radicata*	
32	*Juncus articulatus*	wet site
33	*Juncus* cf *tenuis*	wet site
34	*Linum catharticum*	sward
40	*Mimulus guttatus*	wet site
46	*Prunella vulgaris*	wet shaded sward
48	*Raoulia* spp	open shingle
56	*Sanguisorba minor*	
57	*Sedum acre*	open site on sand/shingle
82	*Epilobium rostratum*	
103	*Isolepis setacea*	wet sward
104	*Rhytidosperma maculatum*	
105	*Carex* sp 1	wet site
110	*Oxalis* cf *corniculata*	
500	Unid. Bryophytes	

Table 4.11.1. Invertebrates consumed by ferrets and cats in the Mackenzie Basin. (Data courtesy of Amelia Pascoe). * = found in the present study

Invertebrate identification	Number eaten
Dermestes beetle larvae	111
Hemideina maori weta	35 *
Hemiandrus new sp. weta	34 *
Hexathele sp. spider	6
Fly maggot	6 *
Noctuidae moth caterpillar	6 *
Aquatic caddisfly larvae	6
Fly adult	4 *
Phaulacridium marginale grasshopper	3 *
Agelenidae spider	3 *
Monomorium antarcticum ant	3 *
Potamopyrgus antipodarum aquatic snail	3
Prodontria new sp. grassgrub beetle	2
Conocephalus sp. grasshopper	2
Histeridae beetle	2
Noctuidae moth adult	2 *
Eumenidae beetle	1
Sigaus minutus grasshopper	1 *
Costelytra zealandica grassgrub beetle	1 *
Metaglymma tersatum Carabidae beetle	1 *
Chrysolina sp. St Johns Wort beetle	1
Dermestes beetle adult	1
Linyphiidae spider	1 *
Solitary hunting wasp	1
Forficula auricularia earwig	1 *
Lygaeidae bug	1 *

4.12. Invertebrates of Conservation value

Several undescribed species have been found from the samples within the Tekapo and Ohau riverbeds. Specimens were obtained from the two samples analyzed for the purpose of this thesis, and a visual inspection of those traps which were not analyzed. Where appropriate, the impact of bulldozing on them is assessed.

***Hemiandrus* new sp.** (Orthoptera: Anostostomatidae) (Figure 4.12.1). This species of burrowing ground weta was first found in the site 20 in the December sample (not analyzed for the purpose of this thesis). The impact of bulldozing its abundance is shown in the previous section; this weta was not recorded in the Experimental Shingle habitat after bulldozing (see Table 4.11.2). It is also one of the two weta consumed by cats and ferrets (see Ryan, (1994) and Pascoe, (1995)).

***Prodontria* new sp.** (Coleoptera: Scaraboidea) (Figure 4.12.2). This grass grub beetle was first recorded from site 23 (a grassy area in the Control Willow habitat) in the December sample. The beetle is winged which is a feature of the genus *Odontria*, but the wing is short and narrow (stenopterous) and the beetle also has a mesothorax characteristic of the wingless *Prodontria* (Peter Johns, pers. comm.). A revision of the distinctions between these two genera is required. The species was not recorded from the two summer samples analyzed, but specimens have been found in the spring and autumn samples (not analyzed). A few specimens were recorded from the cat and ferret guts.

***Scatella* new sp.** (Diptera: Ephydridae). This species appears in the open areas of braided riverbed. It has macropterous (winged) and stenopterous (reduced wings, flightless) morphs. The discovery of this genus in a riverbed environment represents a new locality. It is intermediate in wing patterning between the species *S. nelsoni* and *S. abbreviata.* This species is likely to be the species classified as *Aneuria* new sp. in an unpublished report prepared for the Department of Conservation by Peter Johns concerning the impact of herbicide on terrestrial invertebrates of the Ahuriri riverbed. Although the species was recorded from a number of open shingle habitats, only the winged (dispersible) morphs appeared in

the Experimental areas after bulldozing (pers. obs.).

***Actenonyx* new sp.** (Coleoptera: Carabidae). This species was also recorded in the Ahuriri survey (see above) although due to time constraints, it was not separated from *Actenonyx bembidioides* in the present study.

***Metaglymma* cf *tersatum*.** (Coleoptera: Carabidae). A few members of this genus are found in the stony soils of Canterbury and Otago. The few (< 5) specimens taken from Control and Experimental Willow habitats indicate that the reinstatement of the species *tersatum* from the Cantabrian species *aberrans* may be necessary (Peter Johns, pers. comm.). Because so few specimens were found in the present survey, nothing could be inferred about the impact of bulldozing on this insect.

Several of the wasps (Hymenoptera) found in the present study (identified by Mr. John Early, Auck. Inst. Mus.) could not be identified to species because of a paucity of information on the families in New Zealand, or the time and number of specimens required to identify them accurately. Two records are presented here.

***Scelio* sp.** (Scelionoidea: Scelionidae). This species was found in the 1994 sample of site 24 (Control Willow). Not many specimens of this genus have been collected in New Zealand, and the specimens known to John Early are from subalpine zones where they probably parasitise eggs of mountain acridid grasshoppers. From its (large) size, *Phaulacridium marginale* is too small a host and *Brachaspis robustus* is more likely.

***Microtelenomus* sp.** (Scelionoidea: Scelionidae) Only one specimen was found of this species, from the 1994 sample of site 20 (Control Shingle). *Microtelenomus* are parasitoids of spider eggs; it is an Australian genus and this specimen represented the first record of that genus in New Zealand.

Two Orthoptera have previously been recorded from the Ohau-Tekapo area and already have recognised conservation value.

Brachaspis robustus (Orthoptera: Acrididae). One single specimen of this species was caught in the area which was not included in the analysis and less than 10 specimens were observed in the Ohau and Tekapo riverbed area. Most were seen along the edges of Tekapo flood channels cleared by the County Council in 1985. None were seen in the bulldozed areas before or after treatment.

Sigaus minutus (Orthoptera: Acrididae). Three specimens of this were found in the Control habitats only, although specimens were caught in the unanalyzed samples from the Experimental Shingle sites. Due to the small number of specimens recorded from the Tekapo and Ohau habitats, the impact of bulldozing on them could not be assessed.

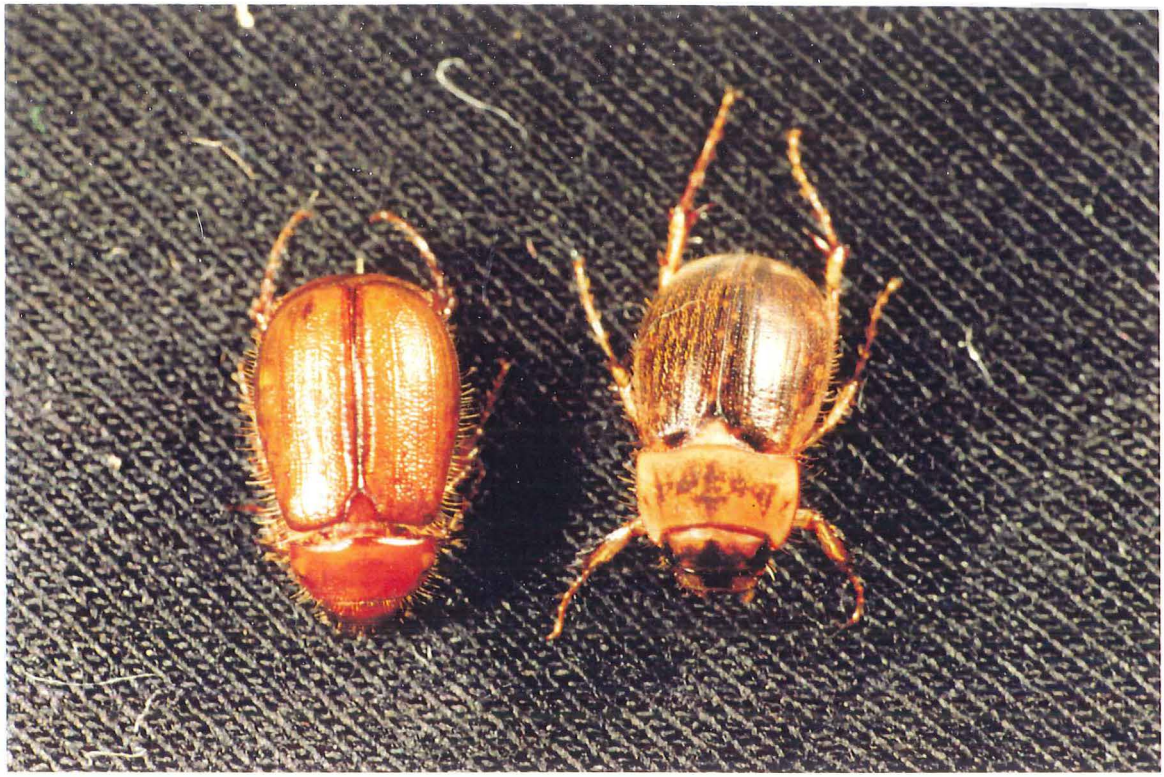

Figure 4.12.2. *Prodontria* new sp. (right) next to the common grass grub beetle *Costelytra zealandica* (left). (actual size about 1.5 cm long). Photo courtesy of Ed Walls.

Figure 4.12.1. *Hemiandrus* new sp. (actual size about 2 cm long). Photo courtesy of Ed Walls.

Chapter 5
Discussion and recommendations

DETECTING AN IMPACT AT SITE, HABITAT AND COMMUNITY LEVEL.

Using simple methods

Species richness, evenness and diversity.

The effect of bulldozing increased the number of species found in the Experimental habitat. This was especially noticeable in the (initially species-poor) Shingle habitat. This increase in taxa contrasts with several other studies which showed decreases in invertebrate richness following disturbance (e.g. Ditlhago *et al.*, 1992), but this may be due to the time at which the invertebrate communities were observed after disturbance, and how quickly the communities recovered.

Morris (1979) found that species responses to disturbance were not uniform, and some species recovered faster because of their life cycle characteristic (e.g. having a double brood per year), they had better powers of dispersal. Invertebrate species richness, diversity and abundance decreased after cutting of calcareous grassland, but the effect depended on the time of year and the level of treatment (Morris, 1979; Morris and Lakhani, 1979). Ditlhago et al., (1992) noted a decrease in abundance of few invertebrate taxa immediately after reed bed management, but the effects on richness, evenness and diversity were barely detectable one year after treatment. Good and Giller (1991) noted that disturbance by frequent cultivation reduced rove beetle (Staphylinidae) populations but once a crop layer had re-established, the recolonisation was rapid.

Invertebrate richness has often been found to be higher in botanically rich environments (Margules and Usher, 1981; Moeed and Meads, 1992). In the present study, however, plant diversity alone does not explain the greater invertebrate richness in willowed habitats. This is reflected by the sites which had a significant change in vegetative richness from 1993 to 1994 (sites 3 and 18) but significant

changes in invertebrate richness were not recorded from the same sites (sites 10 and 14). In contrast, Quinn *et al.*, (1991) found no relationship between plant richness and invertebrate richness.

In the present study, invertebrate richness seemed to reflect botanical richness in areas which only contained very few plant taxa (e.g. sites 1 and 2 of the Ohau). Southwood *et al.*, (1979) found a similar trend between invertebrates with plant succession in that invertebrate diversity and richness reflected plant richness in the early stages of succession, but in later stages factors other than botanical richness became important in explaining invertebrate diversity. Fielding and Brusven (1993) noted that grasshopper numbers were positively correlated with plant species richness, but that it was not a directly related function. Nilsson *et al.*, (1988) concluded that simple single-factor explanations for variations in carabid richness were of limited value.

Alternatively, invertebrate richness has been related to the structural diversity of an environment. A few authors have noted greater numbers of species in wooded areas (Southwood *et al.*, 1979; Webb, 1988; Chandler and Peck, 1992). For example, Uetz (1979) found that litter provided a vertically complex habitat with many microsites, which enabled rare species to persist in areas which would otherwise be dominated by a few competitively superior taxa.

However, Luff and Eyre (1988) suggested that species richness was related to trap accessibility rather than plant richness. Traps in open sites caught more species than traps surrounded by vegetation because plant cover impeded trap access. If trap access was sufficiently impeded by vegetation in the present study, then (the species richness being equal) a higher catch of species would have been expected in shingle habitats. However, in the present study, the species richness was lower in shingle habitats than willow habitats, and therefore relatively low richness in shingle areas probably reflects the true paucity of species found in shingle environments.

Changes in richness will be less obvious in sites where the number of taxa is similar before and after disturbance, but there is a change in species composition.

Also, taxa which are lumped into categories (e.g. Collembola in the present study) will under-represent species richness. In contrast, the inclusion in the richness analysis of 'rare' species (where stochasticity determines presence or absence) could have profound effects on species richness scores. Good and Giller (1991) did not include any species which were represented by less than 3 individuals as they were likely to represent 'tourist' species (non-predatory species which had no lasting or ultimate associations with the habitat). However, Moran and Southwood (1982) argued that tourist species could be consumed by permanent predators and therefore become part of the community trophic structure. The decision to exclude organisms from the analysis quickly becomes one of where to stop! Any index which uses an application of the richness score will also be affected by these problems.

Despite an increase in species richness in the Experimental habitats after treatment, there was no major effect of bulldozing on equitability. The slight increase in evenness was predominantly due to a decrease in abundance of 'small grey' Collembola.

The change in dominance of mites and Collembola may be due to a real treatment effect. Malinda *et al.*, (1982) reported that mite and Collembola populations were at half their pre-trial levels even a year after application of herbicide treatment; this resulted in a long-term change in abundance from a Poduridae dominated soil fauna to one dominated by Entomobryoidea. However, the change in equitability in the Control Willow habitat masked any increases in the Experimental habitats that may have resulted from treatment. Johns (unpublished Ahuriri report for DoC) also noted Collembola capture and hence abundance was influenced by weather conditions.

Highly abundant organisms also affected invertebrate diversity when Margalef's index was applied to the site and habitat data, (although the effect of huge Collembola abundance was downweighted through the logarithmic transformation). Because the trends were very similar to those of species richness, the extra effort required to obtain the components of diversity, (in this study at least) appears to have been unjustified.

Clearly, these dominance and diversity indices are biased towards minute species which are highly mobile, have multiple generations with direct development (hemimetabolous), while species richness scores are biased by taxonomic distinctiveness, and chance-appearance rare species. In samples where a range of different sized taxa occur, taxonomic identification is uncertain, and the number of rare species is high, richness, evenness and diversity indices have limited value.

Relative number of taxa between habitats and their persistence between samples.

Even from a measure as simple as the number of species represented across the five major orders, it is clear that there is a huge amount of variation within and between habitats. What is also evident is that (from this study at least) no highly distinctive patterns of taxonomic representation exist within any given habitat. This may be partly caused by the relatively small sample sizes, or a seasonal influence from flood-delayed sampling. Trapping for a longer period or trapping in more areas may ameliorate the sample size effect, although it should be noted that the number of rare species will increase with sampling effort (Magurran, 1988).

The variability within and between habitats was also emphasised by the persistence data. A striking feature of this data was the low (< 50 %) persistence of invertebrates even in the Control Willow habitats. The suggestion by Uetz (1979) that a litter layer provided an insulating effect on the soil environment does not seem to have validity in the present study. Other authors have found similar variability from samples taken in the same location. Weatherley and Ormerod (1990) examined constancy of invertebrate communities in manipulated and unmanipulated streams. They observed that there was evidence of increasing dissimilarity with time between faunas at the same location, possibly as part of long-term cyclic changes, or random events in the short-term such as immigration and emigration.

However, it was expected that the plant community at least would be more static, but this proved not to be the case. The Control Willow habitat retained the highest percentage of species from 1993 to 1994, and it was surprising that the

Control Shingle habitat retained less; one possible explanation for this difference may be that Shingle habitats were more exposed and subject therefore to environmental extremes. Quinn et al.,(1991) suggested that pre-treatment communities with high diversities tended to have similar species compositions in subsequent samples when compared with the less diverse communities. Scott *et al.*, (1988) observed that tussock grasslands in the Waimakariri River basin showed relatively large variability between years and between sample sites. Some of this variability (in the present study at least) could be related to field error through observer bias or 'blind spots' (Scott, 1989). This seems unlikely to be the main cause of variability in the present study because pitfall traps and quadrats were replicated, and groups of traps rather than individual ones were taken as representative samples.

Without a prior knowledge of a disturbance, it would be difficult to separate habitats adequately using both these measures. Additionally, supposing that the change in relative proportions of the invertebrate orders or the persistence of taxa was related to disturbance, there could be no way of knowing whether the starting point was 'typical' because there has only been one investigation prior to disturbance; the pre-treatment sample may itself be a short term aberation in a period of long-term stability.

Environmental data.

The main impact of bulldozing was on the loss of leaf litter and a reduction in shade in the Willow habitat, and a decrease in lichen cover in the Shingle habitat. An increase in bare ground was also recorded from both habitats. An implication from this is that if any one of these variables were to be measured in isolation then only a partial separation of sites would result. Habitats varied in suites of environmental variables, and no isolated factor could be used to distinguish between every habitat type. Despite this, an effect of bulldozing on the (vegetative) environment at least would be more apparent using the macro-vegetation scores and the major environmental variables than any of the techniques discussed so far. (Whether the impact would be as apparent at a site level is unknown). However, the

caution regarding the position of these fluctuations within a long-term cycle also applies here.

What is most evident from the environmental data is that there is a higher % cover of vegetation on the post-treatment Experimental and Recovering Treatment habitats than there is in the Ohau habitat, even after bulldozing and herbicide application.

The measurement of environmental variables alone is of no use unless they are put into an invertebrate context. It is always possible that the environmental variables measured do not correspond with factors which are important in determining invertebrate assemblages. (This becomes more apparent when environmental data are included in ordination techniques (see below)). Also, factors affecting some species may not elicit a response in other species. Weatherley and Ormerod (1990) observed not all invertebrates responded to a changed environment in a similar manner; a few species of invertebrate exhibited change in abundance and distribution with a changing environment, but the majority were resilient to perturbations and showed considerable inertia to perturbations in the short term. Joern and Pruess (1986) found that only those invertebrates in a similar trophic position, which fed on the same types of food in the same way, responded to disturbance in a like manner.

Using complex methods

Community classification.

In general, the groupings of sites according to the plant and invertebrate classifications were similar. Both separated the bare shingle Ohau site 1 from the other sites which were vegetated, and the shingle habitats from the willow habitats and both from the bulldozed sites (in the invertebrate classification at least).

Several observations can be made regarding the relation of the invertebrate

groupings to the plant grouping. If invertebrate assemblages were related purely to plant composition, then it could be expected that the sites would conform to the pattern of sites based purely on the plant communities. However, the site separation based on invertebrates does not exactly follow the plant classification pattern; several observations can be made from this.

1) In the invertebrate classification, site 2 (Ohau) was grouped with other semi-vegetated areas; this may imply that invertebrate communities reflect the amount of vegetation cover rather than the actual composition of the plant community.

2) The flooded sample of site 3 in 1994 was grouped with some of the bulldozed and Recovering Treatment sites in the invertebrate classification; this could imply that (large scale) disturbance is detectable in invertebrate communities, irrespective of causal nature of the disturbance (bulldozing or flooding).

3) In the plant classification, the inclusion of site 16 (pre-treatment Experimental Willow) with the Recovering Treatment sites was unexpected, as the site was grouped along with all the other 'willow' sites in the invertebrate classification. Three possible explanations are given for this. Firstly, the initial site label as 'willow' may have been inappropriate. However, this seems unlikely as the traps were set directly under willow trees amongst the litter, and the area appeared visually to be as typical of a 'willow' site as any other within the Willow habitats. Secondly, plant classification may reflect a real (but invisible) similarity between this particular site and the Recovering Treatment sites, and thirdly, invertebrates may be detecting real dissimilarities between these sites.

The question which then arises is 'which classification (plant or invertebrate) is more real or correct' and the answer to that, of course, is both. Clifford and Stephenson (1975) were justified in saying that there is an ever present risk that the boundaries of the species will not fit with where humans decided the taxa should be.

It must be mentioned however, that if the invertebrate communities were to be predicted from the groups of sites based on the plant classification, then some (but not the majority) of the invertebrate habitats would have been predicted incorrectly.

The risks inherent with incorrect predictions of invertebrate habitats must be recognised by the institutions using these results. Eyre and Rushton (1989) advised that classification (and ordination) of invertebrate habitats in complement with plant data should be practised to provide an ecological basis for the measurement of invertebrate 'typicalness' (of a habitat) for conservation purposes.

Not surprisingly, the use of classification techniques has received recommendation as well as rejection. For example, Hutcheson (1990) found that diversity indices alone did not adequately group samples by site whereas classification distinguished the two 'clearly different' communities, summarised the information in an ecologically meaningful manner and overcame large temporal and spatial variation in his data. In contrast to Hutcheson (but similar to the present study), Hubbard and Bastow Wilson (1988) found that the most striking feature of their classification of plants in the Clutha district was the lack of clear structure in the species by site (two-way) classification table; the predominance of intermediate values showed that no group of species was reliably present or absent from specifiable vegetation types, and the general lack of constant or faithful species was another indicator of this.

Despite this, what is evident from both the plant and invertebrate classifications is that neither the bulldozed Experimental sites nor the Recovering Treatment sites have approached the Ohau state significantly enough for all these sites to be grouped with the Ohau communities; neither have they reverted to either the stable shingle or wooded communities even two years after treatment.

Community ordination.

In general, ordination of invertebrates and plants distinguished shingle sites from bulldozed ones and from willow habitats also. Even though the bulldozed and Recovering Treatment sites shifted away from their original positions in the ordination diagrams towards the Ohau, the Ohau sites still remained reasonably distinct (similar to the classification). What is also evident from the ordination (and in the classification) is that the bulldozed sites do not resemble either shingle or

willow habitats even two years after treatment. Indeed, from the plant community ordination, the Recovering Treatment habitat appears to have stabilised (within the fluctuation limits seen in the Control Shingle and Willow) in an intermediate position. The position of the Experimental Willow sites on the plant ordination in relation to the Control Willow sites tends to affirm the suspicion generated from the plant classification that the initial site choice was not very consistent. One striking feature of both the plant and invertebrate ordinations is the variation of sites within the Control habitats, both between the sites (within a habitat) and between years. Some of this is likely to be a seasonal effect due to the flood-delayed sampling.

Ordination techniques which use abundance data will be subject to the same problems which occur where abundance data is used in evenness and diversity measures (although the transformation of the data down weights this effect). Quinn et al., (1991) carried out separate ordination analyses on abundance and incidence data respectively to avoid potential problems with interpreting abundance data from pitfall traps. They found that both abundance and incidence data produced similar patterns. Again, this implies that (for some community studies at least) the extra effort required to obtain abundance data may be unnecessary. Also, the decision whether to include rare species also applies here; species which appear as a seasonal effect, and/or rare species (especially ones which are only found in one trap) tend to be outliers on the species ordination graphs, and contribute little more to the species ordination diagrams than an extra dot to be numbered! In contrast, ubiquitous species tend to blur the distinctions between separate clusters, but by removing them the researcher is probably left with an assemblage of habitat specific species. This achieves the purpose of distinguishing between species assemblages, but without the need to perform an ordination! The use of incidence data would not overcome this problem.

One obvious hazard in interpreting the invertebrate and plant ordinations, is the fact that the extraction of axes appears to be related to different gradients (namely the moisture gradient for plants contrasted to the disturbance gradient in the invertebrate ordination). This implies that *the factor which distinguishes between the*

plant communities is not necessarily the same factor distinguishing the invertebrate communities.

However, finding explanations of the underlying gradients may not be necessary. Fielding and Brunsven (1993) could not identify the underlying processes which generated the observed ordination patterns, but they concluded that regardless of the mechanisms involved, the differences between disturbed and undisturbed sites were distinctive.

Similar to Hubbard and Bastow Wilson (1988), all the invertebrate species ordinations in the present study showed a lack of structure; (species were not arranged in a narrow, dense band across the diagonal which would be expected if the variation in species were well summarised by the initial few components). These authors concluded that the inability of ordination (and classification) to reveal clear structure suggested that there was no simple gradient present which could explain (invertebrate) community variability. This conclusion can also be applied to the present study.

In terms of the amount of variance explained, the distinction between sites based on plant ordination was better than the invertebrate ordination. This is not unusual. For example, Fielding and Brusven (1993) used DCA of vegetation which confirmed the validity of initial classification of sites by a clear separation amongst the hypothesised areas, but the separation was less clear using grasshopper assemblages than using plants alone. However, in the present study, the amount of variance explained in the ordination with plant species alone was less than that in other studies. Cowie et al., (1992) found that the major axis accounted for 88 % of the variation in the data.

When plants and environmental data were included as a secondary matrix, the amount of variance explained by their inclusion was similar to that explained in the ordination of invertebrate data alone. The amount of invertebrate variance explained when plants were included in this present study was less than that found by other researchers. For example, Fielding and Brunsven (1993) found that the inclusion of

vegetation variables as a secondary matrix accounted for 60-70 % of the total variation in the grasshopper community. The amount of invertebrate variance explained when environmental variables are included in the ordination is similar to that found by other researchers. McCune and Allen (1985 in Hubbard and Bastow Wilson 1988) found that only 10 % of the variation in some Montana forests could be accounted for by the measured environmental variables. One advantage of constraining invertebrate (or plant) data with environmental variables is that the biplot gives an indication of the (measured) factors which separate species, and hence habitats.

From this study, the distinction of sites based on plant data alone explains more variation between sites than does the invertebrate ordination, even when plant and environmental data are included as a secondary matrix. However, care should be taken if plant data does not group sites in the 'pre-determined' or expected manner; the same risks of wrongly predicting an invertebrate environment using classification techniques also apply here.

RECOMMENDATION

The use of simple community measures to detect a treatment effect is not recommended, although species richness could be obtained with minimal effort. The use of more complex measures is advised as they rely less heavily on aspects of the data which are affected by trap bias or small sample sizes. Classification and ordination techniques of both plants and invertebrates provided the best overviews of the community data. The inclusion of plant species and environmental variable measures is recommended to quantify (at least partially) the invertebrate environment and to justify the initial site and habitat distinctions.

DETECTING AN IMPACT AT SPECIES LEVEL.

Change in species composition towards the Ohau state and loss of taxa from bulldozed areas.

Braided riverbed invertebrate taxa which occurred in the treatment areas after bulldozing were represented by only a few specimens. What was surprising was the appearance of flightless species (e.g. Bethylidae) after disturbance. Several authors report immigrant species as having good powers of dispersal (e.g. den Boer, 1987; Hutcheson, 1992). The presence of this (rarely sampled) species could indicate that perhaps the soil conditions were not disrupted evenly across the whole Experimental habitat.

What becomes apparent is that even two years after bulldozing, the flora and fauna of the Experimental and Recovering Treatment habitats has not significantly approached that of the Ohau habitat (despite frequent herbicide applications).

One question which arises from this study is how much is an improvement and what would an improvement actually represent. Should an improvement be measured by abundance or merely presence? It was clear that in bulldozed habitats, *Actenonyx* were not found in the same abundances as thos which occurred in the Ohau. The majority of these beetles in the Ohau occurred in Ohau site 1 (unvegetated shingle), and clearly, the bulldozed habitats do not approach this state. Based on presence however, it appears that there have been increases in braided riverbed taxa, but these this may be caused by factors other than a permanent change in habitat suitability. For example, both the Carabidae (*Actenonyx bembidioides*, *Bembidion* sp 1, *Notogonum feredayi* and *Hypharpax* sp 1) are strong fliers and their presence is probably due to the flooding in the Tekapo which temporarily created a suitable habitat. (It is also possible that presence of Ohau species in the bulldozed areas could be chance related as they were not strongly represented in either the Ohau or the Experimental habitats).

It is interesting to note that *Bembidion* are often found in riverbank habitats

(Anderson, 1985). However, their presence in the pasture-like site 3 of the Ohau indicates that they occur in vegetation *regardless* of whether or not it is native. (The adaptation of species to human-modified environments is not unusual; *Costelytra zealandica* is a prime example (Moeed and Meads, 1992)). Several of the invertebrate taxa recorded after bulldozing occur in a suitable habitat rather than a braided riverbed habitat *per se*. For example, the introduced Ladybird beetle and the lacewings both consume aphids (themselves introduced) (Hodek, 1973) and their appearance reflects the abundant aphid food supply on a weed infested environment, rather than a typical braided riverbed habitat.

Chandler and Peck (1992) found that generalist insects were unaffected by disturbance, but species with specific (tree-related) habitats and specific food requirements reduced in abundance. Many of the species lost from the Experimental areas either required stable soil in which to burrow or had specific habitat requirements, had poor powers of dispersal or were poorly represented. The now species of *Hemiandrus* has all of these characteristics.

The large number of ubiquitous invertebrate taxa reflects the generalist nature of many of the sampled taxa. Many of them were either introduced or native pest species, had good dispersal ability, or were in lumped categories due to their small size or taxonomic indistinction. *Priocnemis nitidiventris* was one native species which was perhaps favoured by clearance, as bulldozing created a large amount of loose silt in which the adult burrows to deposit a spider upon which the larva feeds.

This opposite phenomenon is reflected in the plant taxa; very few were native pioneers; all 'braided riverbed' plant taxa appearing in the bulldozed areas were recorded from the lush pasture-like site 3 of the Ohau. Species typical of braided riverbeds (and found in site 2 of the Ohau) include *Epilobium melanocaulon*, *Raoulia* spp, and *Elymus rectisetus* (Burrows, 1977), none of which appeared in the bulldozed sites. The lack of pioneering species on the bulldozed sites probably reflects inappropriate conditions for these species and the time at which the samples were taken, rather than their dispersal ability (but see Ash *et al.*, 1994). Similar to the invertebrates however, the majority of the plant taxa which disappeared from the

Experimental habitats after bulldozing occurred in specific habitat types, associated with shaded or vegetated (often wet) areas.

Invertebrates as Wrybill and Dotterel food

Bembidion were amongst the items of Wrybill prey mentioned by Pierce (1979). Although these have not increased significantly in the Experimental habitats following bulldozing, they occurred more frequently in the Recovering Treatment habitat in 1994. However, the sites sampled in the Recovering Treatment and Experimental habitats were not riparian sites, so whether this increase is indicative of an increase in food availability or was merely a short-term aberation can only be assessed with further study. No recommendation about the suitability of bulldozed areas for these birds can be made based on the invertebrate species data from this study.

Invertebrates as predator food

From the small numbers of weta caught in this study, nothing concrete can be said about the effect of bulldozing on increasing or decreasing this supplementary food source for predators, and nothing can be implied about the consequent predation pressure on riverine birds.

Even if bulldozing had decreased the availability of invertebrate prey, it is likely that most predators would be unaffected because of their ability to cover large amounts of territory or colonise new areas and therefore increase their chance of encountering vertebrate and invertebrate prey (King and McMillan, 1982; Taylor and Tilley, 1984; Fitzgerald and Karl, 1986; Murphy and Dowding, 1994).

Although invertebrates were consumed by over half the predators, invertrebrates represented a minor part of ferret and cat diet. Several authors investigated the weight of prey items consumed and found that invertebrate prey contributes less than five percent of the total diet by weight. For example, Langham (1990) found that mammals (mostly rodents) were the staple food for cats

by weight of prey consumed contributing 74 % to the diet while invertebrates contributed less than two percent. Catling (1988) observed that although invertebrates occurred in about 50 % of the cat guts, they contributed less than two percent by weight.

Alternatively, invertebrates may be consumed for purposes other than calorific gain. Ryan (1994) found that invertebrates contributed only 0.06 Kilocalories per invertebrate meal, while Konecny (1987) suggested that invertebrates had a higher water content than vertebrates and were therefore consumed more often in the dry season.

However, the importance of invertebrates as a food source for some predators should not be dismissed. Although invertebrates may not constitute a large part of predator diet, reliance on alternative invertebrate prey can vary between individual predators, as well as seasonally and with the age of the predator.

Fitzgerald and Karl (1991) found that although invertebrates were consumed infrequently, two cat guts contained large numbers of emerging dragonflies (Odonata). Of the 33 scats collected by Langham (1990), 11 contained only cicada nymphs (Homoptera: Cicadidae) and constituted an important part of the diet when they were seasonally abundant. Catling (1988) found that juvenile cats changed from young rabbit prey in winter and spring to a supplementary diet of invertebrates in summer and autumn when only adult rabbits were available.

Gibb *et al.*, (1969) observed that cats consumed a greater variety of food in autumn when young rabbits were scarce. King and Moody (1982) found that female mustelids ate more small prey than males, and that juveniles consumed seven times the amount of invertebrate prey than adults. Ryan (1994) studied cat diet in the Mackenzie basin and observed that eleven species of invertebrates were consumed and large invertebrate prey were weta (the new species of *Hemiandrus*, and *Hemideina maori*) and *Hexathele* spiders. Of the 40 *Hemiandrus* consumed, 28 were was consumed by one cat. Fitzgerald and Karl (1979) found that rats were the most important food for cats in Orongorongo valley, and invertebrates (mostly weta) were

eaten frequently but dragonfly and cicada were consumed when seasonally available; all invertebrates contributed less than two percent to the diet by weight.
Fitzgerald and Veitch (1985) found that weta were eaten by 42 % percent of the cats sampled on Herekopare Island, and seven cat guts contained more than ten weta while two contained more than 100 weta. They concluded that "when taken in such numbers these insects can make an important contribution to the diet of cats" (page 322).

RECOMMENDATION

It seems unlikely that a change in the availability of invertebrate prey would affect the majority of predators, but sample sizes in the present study were too low to determine an effect of bulldozing on the main invertebrate prey (Orthoptera). Some predators relied heavily on invertebrates as an alternative food source. Therefore, hunting or trapping specifically for these invertebrates is advised to obtain a better understanding of their availability.

Invertebrates of conservation value

Several of the taxa which have significant conservation value are dispersal-limited on account of their (relatively) small size and inability to fly or cross water. Hopkins and Webb, (1984) stated that extinctions may not be followed by immediate recolonisation as species are not and cannot be assumed to be equivalent in their powers of dispersal. Morris (1979) found that colonisation of bare ground by Heteroptera was slowed by species with poor powers of dispersal and otherwise potential colonisers were often brachypterous. Morris (1979) observed that cutting of grassland resulted in a decrease in the number of individuals, but did not result in elimination of species because immigration occurred.

Three species are discussed below.

If large populations of *Prodontria* had been present in the bulldozed areas, they are likely to have been affected. All grassgrubs are root feeders and the adults

feed on vegetation at the ground surface. Loss of vegetation due to bulldozing and disruption of the soil horizons would severely affect this species. Being flightless it is dispersal limited, and could not rapidly recolonise the area. However, the beetle appears to have survived in human-induced grassland and has been caught found in similar grassy areas upstream (F. van Wyngaarden, pers. comm.). More research is required to classify this species and determine its ecology (e.g. whether it is localised in distribution).

If large populations of *Hemiandrus* had been found in the Experimental area, they are also likely to have been affected by the destruction of their burrows and habitat in compact fine silt (Frans van Wyngaarden, pers. comm.). Subsequent to the discovery of this species, its ecology and distribution were investigated by Frans van Wyngaarden (unpubl., 1995). Like the *Prodontria* sp, it does not appear to be restricted to the Tekapo delta.

Brachaspis robustus is a grasshopper listed as an 'A' category species which should receive the highest conservation priority (Molloy and Davis, 1992; Tisdall, 1994). It is extremely habitat specific and is known only from stony floodplain terraces and braided riverbeds within the Mackenzie basin (Bigelow, 1967; White, 1994). For this reason, it appears to have extreme value as an indicator species. Although none was sampled in the present study, had this grasshopper occurred in the Experimental areas after treatment, its presence would have indicated a significant improvement in habitat quality towards the Ohau braided state.

However, even if the bulldozed areas had represented suitable habitat, the presumption that species will immigrate into a suitable area after a disturbance is unrealistic. Most of the invertebrates in this study which have conservation significance are dispersal limited. White (1994) reported that nine weeks after a flood event, *Brachaspis robustus* had dispersed no further than 21 metres, and the majority moved far less. Assuming a suitable habitat is available, transferral of specimens or captive breeding and release of these grasshoppers would be the only method of expanding their distribution.

RECOMMENDATION

All the invertebrates of conservation value were caught in small numbers and little could be deduced about the impact of bulldozing on them. None of the sampled taxa is likely to be threatened with immediate extinction from bulldozing, although the majority were affected in the short term. However, **targeted trapping for these species is advisable** to monitor their populations in areas before restoration work occurs. Some delay will be necessary to allow the prospective researcher time to accumulate and evaluate results and identify further new species which are likely to appear). Translocation of and/or Captive breeding and release may be the only method for expanding the distribution of dispersal-limited or rare taxa.

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Appendix 1

Weather observations for the two sample dates.

16-19 January, 1993: fine, warm and breezy at times, otherwise still and muggy. Intermittent rain 18th, steady rain in evening. Fine and warm the following day.

11-14 February, 1994: light breezes and patches of sun showers,hot. High cloud in afternoons. Overcast and cool on morning of 14th.

Appendix 2
Pitfall traps

Pitfall trapping has been used extensively to study the distribution and abundance of surface-active arthropods especially carabid beetles (Ahearn, 1971; Halsall and Wratten, 1988; Topping and Sunderland, 1992). Pitfall traps consist of a container sunk into the ground with the lip usually set flush with the ground surface. Invertebrates are captured when they fall into the container and are unable to escape (Southwood, 1978).

Despite the widespread use of pitfalls, the technique has been heavily criticised for not providing a measure of the true abundance, distribution, and life history of single species or of invertebrate communities. The discrepancy between trap catch and observed communities results from two main factors. These are the characteristics of the trap technique used, and the characteristics of the invertebrates themselves.

Characteristics of the trap technique.

Pitfall traps can be 'live' or 'kill' traps. Live traps are dry containers which retain invertebrate species because they are unable to escape mechanically: the sides of the container are too difficult for the animal to climb, or a substrate is placed in the the container to encourage the organisms to remain passive. However, the traps may not collect representative samples of the community if captured organisms release pheromones which repel or attract others to that trap (Ahearn, 1971), or the invertebrates are taken by predators such as birds or other invertebrates also captured in the trap (Baars, 1979). Bird predation can be overcome by providing a roof to the trap, but this introduces a bias in that the roof can itself deter some invertebrates. For example, Baars (1979) found that some beetles were caught more frequently in traps with wire mesh roofs than in traps with a solid metal roof.

Halsall and Wratten (1988) investigated the efficiency of pitfall traps by recording (on video tape) the number of encounters of a given number of carabid

beetles with the edge of a dry-trap that resulted in capture. One species was able to walk around the walls of the trap and individuals were 'captured' by remaining passive at the bottom! Climbing species would be under represented in field samples. One way of reducing this bias would be to use smooth sided glass containers.

Predation by other trapped species and loss of climbing and volant species can be minimised by including a liquid (usually a toxin) in the trap so that trapped invertebrates die before they can escape (hence 'kill' trap). However, some killing solutions (e.g. formalin) can be detected by some invertebrates and avoided, and others can be attractants (e.g. the spirit preservatives used by Greenslade and Greenslade (1971)). Ethylene glycol is commonly used in pitfall traps and has the advantage that it is odourless (Duffey, 1962), can withstand some dilution before losing its potency, and does not evaporate quickly. For these reasons, ethylene glycol was used in the present study.

Some authors (e.g. Asquith and Messing, 1992) have included baits in pitfall traps to selectively attract invertebrates rather than relying on passive capture. The use of baits introduces other problems. Greenslade and Greenslade (1971) found that more ants were attracted to traps with a beer-syrup mix rather than just alcohol, but also noted that the bait (and invertebrates within it) decomposed, which increased the attraction of the trap. Kirk (1984) found that adult Phoridae (Diptera) were more abundant in white than yellow traps, but the response varied between species.

The response to the trap may not necessarily be related to the human-perceived trap attractants. Kirk (1984) also noted that other invertebrates responded to darker shades of trap colour because it contrasted with the background ultra violet light emmittance rather than responding to a distinct visual hue. Aquatic insects including hydraphilid beetles may be attracted to bodies of 'water' within pitfall traps, rather than being associated with a terrestrial environment (pers. obs.). Greenslade and Greenslade (1971) concluded that baits may be worthwhile in studies of a single dominant species but that the additional complications introduced by baits did not always justify their use.

Invertebrates and susceptibility to capture.

Many factors influence the susceptibility of an invertebrate species to capture in pitfall traps. Most of the critical literature on pitfalls focuses on the response of various species to the trap and comparisons of the trapped assemblage with the 'actual' community. Several factors have been proposed to account for these discrepancies (Southwood, 1978).

Most authors recognise the importance of population density and invertebrate activity as the major factors affecting susceptibility to capture. Other physical and biological factors include the 'favourability' of the environment including density of vegetation within which the trap is set (Ahearn, 1971; Baars, 1979); the sex, age, and physiological condition of the organism (Hughes, 1955; Greenslade, 1964); temperature and prevailing weather (Williams, 1954); interspecific differences in behaviour, and time of season sampled and phase of the moon (Ahearn, 1971).

Baars (1979) observed that carabids moved about more in unfavourable habitats and moved much smaller distances in favourable habitats to the extent that the catch of beetles in unfavourable habitats represented a higher proportion of the beetle population in that environment. Greenslade (1964) recognised that variation in catches of a species depended on different types of ground cover and the resistance it presented to horizontal movement. Topping and Sunderland (1992) noted that the vegetation surrounding the trap had a species-specific affect on invertebrate density around the trap. Ahearn (1971) suggested that vegetation around the trap reduced beetle mobility and therefore increased the likelihood of their capture in pitfall traps placed near the vegetation. In contrast, Williams (1954) stated that invertebrate locomotory activity was higher in a wooded environment and activity was associated with a litter layer where the species could rest in between periodic bouts of activity. The trees made climatic effects more uniform throughout a diel cycle, which led to differences in capture rate between wooded areas and open areas. Baars (1979) recommended standardizing trap conditions in different environments (e.g. creating an area of bare ground) to reduce the erroneous factors such as differences in density and structure of surrounding vegetation. Greenslade (1964)

concluded that pitfall trapping would only represent relative abundance accurately if activity was constant between species.

Topping and Sunderland (1992) compared the catch of spiders from pitfalls with that from a D-vac suction sample and found that the catch from pitfall traps overestimated the proportion of male spiders present because of mate-searching behaviour. The order of dominance (most abundant species) in the pitfall samples differed from that in the D-vac sample. Topping and Sunderland (1992) also found that a change in linyphiid density could be related to changes in immigration, and pitfall catch underestimated the juvenile component of the population. Hayes (1970) observed that small and gravid female isopods exhibited a behavioural response which made them less 'trappable'. From the frequency of isopods in core samples, he concluded that pitfalls could not be used to accurately study isopod populations.

Life history stages may also influence capture success. Ahearn (1971) found that seasonal differences between pitfall samples resulted from an increase in eclosion of adult beetles which increased the population density and therefore increased their chance of capture. Williams (1954) noted changes in the proportion of transient species (such as caterpillars seeking pupation sites) in samples taken at different dates. Surface activity of Tenebrionid beetles increased with an increase in surface illumination from moonlight (Ahearn, 1971).

Temperature may be a dominant factor determining invertebrate activity if they were poikilothermic and activity was therefore temperature dependent (Williams 1954). Mitchell (1963) observed however, that nocturnal organisms would be affected differently by temperature than day-active species, but assumed that under certain weather conditions a set of traps should catch the same proportion of the available population at each equivalent period. The activity of species which lose water by transpiration through their integument (e.g. harvestmen, millipedes and woodlice) are likely to be affected by relative humidities and temperature (Procter 1966).

In contrast, Topping and Sunderland (1992) stated that differences in capture

may be unrelated to invertebrate activity, but rather related to the invertebrates ability to detect the trap. According to Asquith and Messing (1992), the susceptibility of one species did not necessarily imply the susceptibility of related species, even congenerics. Halsall and Wratten (1988) found that the overall proportion of encounters that resulted in capture was low (less than 44 percent). For some species, the efficiency was much less. Some taxa were able to detect the trap edge and showed avoidance behaviour such as hanging over the trap edge then retreating to skirt around it. Cary (1981) recorded a capture rate below 33 % for the ground weta (*Hemiandrus* sp) when he disguised the trap rim with leaflitter, but even this relatively low efficiency decreased to less than four percent when the metal trap rims were exposed.

Substrate disturbance around the trap can also bias invertebrate capture. Greenslade (1973) recorded a 'digging-in' effect where ants investigated the disturbed area around the pitfall trap (a new feature in their landscape). There was a high initial catch after the traps were set, but the effect could be partially reduced by placing lidded traps in position for a period before they were set. Earlier, Greenslade noted that beetle size could affect their susceptibility to capture (Greenslade, 1964). Small beetles could recover balance and escape from the pitfall trap whereas larger species had more momentum and were more liable to fall in, increasing their capture frequency.

Advantages of pitfalls traps.

The main problem in constructing a biodiversity inventory has been described as a trade off between three aspects of research methodology (Crosby in Cresswell and Veitch, 1995). Methods are either quick, cheap, accurate, or a combination of any two of these with a compromise on the third (unaccounted-for) aspect. Pitfall traps are often used because they are cheap (empty food or drink containers), are easy to operate, many species can be trapped, and large catches often result (Greenslade and Greenslade, 1971; Southwood, 1978; Halsall and Wratten, 1992; Topping and Sunderland, 1992).

Several authors including Greenslade (1964) demonstrated that pitfall trapping cannot properly be used for quantitative assessment of a carabid population for any habitat nor should it be employed to compare the numbers of any one species in different habitats. Some authors advocate that the only appropriate use of pitfalls is for mark-recapture work to study a limited number of species with appropriate behavioural responses to the trap. For a preliminary investigation into a community, the mark-recapture approach is not feasible. (A community is taken to be the fauna captured in pitfall traps and is by no means extensive). Because species frequently interact between members of a group and with other invertebrates, no single species should be considered in isolation (Greenslade 1973). For example, large populations of ants influence other groups inhabiting the soil surface, which may in turn affect the sampling accuracy of other taxonomic groups (Greenslade and Greenslade, 1971).

Every trap technique has advantages as well as disadvantages. Greenslade and Greenslade (1971) found that, although pitfall traps did not provide reliable quantitative data they, did collect members of groups from the litter surface that were not satisfactorily extracted using Tullgren funnels (e.g. Gryllidae, and entomobryid and sminthurid Collembola with long appendages). Good and Giller (1991) suggested that D-vac samples were biased against samples which were extracted in large numbers using Tullgren funnels; this method is totally impractical in a multi-site study due to the large volume of soil which needs to be handled, and it is unlikely that these extra species would add much to the results obtained from pitfall sampling. Gist and Crossley (1973) found that pitfall trapping showed close agreement with estimates of several Arthropoda when compared to hand sorting of quadrats for species abundance and composition.

Alternative methods for quantifying terrestrial invertebrate fauna are not without their own problems. Alternative methods for quantifying terrestrial invertebrates include direct counts of invertebrates within quadrats in the field; these are labour- and time- intensive and often result in habitat destruction as invertebrates may be clumped in favourable areas (Greenslade 1964). Clearly this approach would be inappropriate for smaller, highly mobile organisms such as Collembola, or for repeated measures in the same habitat. Suction samplers are popular, but the

apparatus is expensive and was unavailable for the present study.

It is clear that catches from pitfall trapping will not represent the entire invertebrate community in an area, nor will it represent the relative abundance of every species in the same environment because each species differs in its susceptibility to capture. Nevertheless, pitfall trapping is still widely used as a trap technique. Classifying sites by their pitfall catch is meaningful as long as it is remembered that this classification is not based on the density but rather the catch. Even if site characteristics such as soil type and vegetation alter the catch, even at the same absolute densities of species, then it is still a useful classification (Luff and Eyre, 1988).

Appendix 3
Diversity

Diversity is an elusive and multifaceted concept when a precise definition is sought (Southwood, 1978). When a fauna is sampled, it is often found that a few species are represented by many individuals, and many species are represented by a few individuals (the largest count category is usually one).

There are two major concepts of diversity: alpha diversity (the diversity of species within a habitat or community) and beta diversity (a measure of the rate or extent of change in species along a gradient from one habitat to others) (Southwood, 1978; Moore and Chapman, 1986).

Alpha- and beta-diversity within a community.

Alpha diversity

Alpha-diversity is a compound concept consisting of species richness (S) the number of species in a sample, and species equitability (N) the distribution of individuals amongst the represented species (May 1975, Ludwig and Reynolds 1988, Magurran 1988). A site with many species (but where one species is dominant) will be considered less diverse than a site with the same number of species but where the individuals are apportioned evenly amongst those species. The measured relationship between the number of individuals in a habitat and the number of species represented by those individuals varies and is related to sample size and sampling effort (Southwood 1978).

A variety of ways exist to describe species diversity in samples containing many species and many individuals (Bullock, 1971). Several simple indices have been derived using some combination of S (the number of species) and N (the number of individuals summed over all the species). There are two main types of indices; parametric where species diversity is explained by an underlying model of species-abundance distribution, and non-parametric.

There are three categories of parametric and non-parametric indices; 1) those which explain the species richness alone; 2) those which explain the equitability or evenness (distribution of the species abundances and the contribution of the dominant species to the total abundance); 3) those which consider the relative abundances of the species in proportion to the species count (Pielou, 1975; May, 1975; Southwood, 1978; Magurran, 1988).

The problem faced by ecologists is how to choose the appropriate index and how to interpret it (Didham, 1992). There are so many indices in common use that one could choose the index which gives the 'desired' result (Didham, 1992). Some workers (May, 1975; Southwood 1978; Magurran, 1988) have strongly advocated the use of entire species'-abundance distributions as providing the only sound basis for examining species diversity within the community because it utilizes all the information gathered in a community and is the most complete mathematical description of the data (Magurran, 1988).

Several models have been applied to community data for explaining the species abundance distribution, presented in the form of the curve produced by the logarithm of the species-abundance vs their rank. The four most commonly applied models are the Geometric Series, the Log Series Distribution, the Log normal distribution, and the Broken Stick model (Magurran, 1988). These describe a progression in species abundance distribution from uneven (geometric series) to even (broken stick model) distribution of individuals amongst the species. Field data have shown that the geometric series pattern of species abundance is found primarily in species-poor (and often harsh) environments or in the very early stages of succession (Whittaker, 1970) where a few species dominate the abundance curve. As the conditions ameliorate, species abundance patterns grade through the log series, log normal to the broken stick which represents a more even expression of species abundance (Magurran, 1988) but in real data sets, this distribution is uncommon (May, 1975). May (1981) suggested that the 'equilibrium' communities fit the log-normal and when disturbed, they reverted to the log series. However, this is controversal as several authors have argued that the log series fits data from undisturbed communities and the log normal is a better fit for transitional community

data (Kempton and Taylor, 1974; Lambshead and Platt, 1985). The controversy is largely unresolved although Death (1991) suggested that a community may progress from geometric through log series to log normal as the community becomes stable, but revert to log series as competitive dominance occurs. Wolda (1981) concluded that the log series diversity coefficient was independent of sample size and the diversity of the samples reflected the diversity of the source fauna (but this estimate became less reliable with smaller samples). Also, Southwood (1978) found the parameters of the log normal distribution was less useful than the log series, but even when the fit of the log series was not close, its robustness provided a useful measure of diversity. However, this seems to defeat the purpose of determining how closely community data fits an underlying model.

The description of S:N relationships in terms of the parameters of a model implies that the model is at least approximately applicable (Southwood, 1978). While models provide the fullest description of diversity data, they are dependent on some tedious model-fitting (Magurran, 1988) and were not used in the present study for that reason. In light of this uncertainty surrounding biological vs statistical species abundance models and the applicability of different models in different situations, it is not surprising that many workers have opted to use non-parametric statistical indices to characterise the diversities of communities. The Q statistic is the non parametric equivalent of the log series diversity parameter, but is computationally complex (Didham, 1992).

Several non-parametric indices exist (Magurran, 1988) which do not assume an underlying distribution model. The simplest non-parametric measure of species diversity is the species count or species richness (S) and can only be applied to community samples of equal sizes (Ludwig and Reynolds, 1988). One problem with counting the number of species is the loss of information concerning the abundance data, and the count is strongly susceptible to sample size and effort (Krebs, 1985). Hence a number of indices have been proposed which are independent of sample size. Where sample sizes are equal, a simple count of the number of species is appropriate (Magurran, 1988). In situations where sample sizes are not equal a technique devised by Sanders (1968) called Rarefraction can be used to calculate the

expected number of species. However, a major criticism of rarefraction is that it leads to a great loss of information regarding the relative abundance of species (Hurlbert, 1971). A well known richness index is Margalef's Index, having the great advantage of ease of calculation and relating S to abundance (Didham, 1992; Clifford and Stephenson 1975) and for this reason, it was used in the present study. The formula for this index is

$$D_{Mg} = \frac{(S - 1)}{\ln N}$$

Like S, it is sensitive to sample size (Magurran 1988).

Dominance measures are weighted towards the abundance of the commonest species rather than providing a measure of species richness (Magurran, 1988), and have been criticised for their insensitivity towards rare species (Clifford and Stephenson, 1975). Despite this they are commonly used, and as long as they are recognised as dominance indices *per se*, the results are comparable. Dominance/evenness indices include the commonly used Shannon-Weiner information statistic (sometimes called the Shannon Index or the Shannon-Weaver Index (Southwood, 1978)), and the Simpson Index. The Shannon-Weiner Index is derived from the information theory which tries to predict the outcome of the next individual species drawn randomly from an infinite population. A greater number of individuals or species will increase this score. Other forms of this index (e.g. the Brillouin index) are more appropriate when the population size is limited or the samples are not random (Magurran, 1988; Pielou, 1975) but this is computationally more complex and time consuming. Whittaker (1972) and Kwiatkowska and Symonides (1986) found that this index was influenced by sample size (quadrat size), and Taylor (1978) found the Shannon index wanting in regard to site discrimination, a major function of diversity indices.

One of the best known dominance indices is Simpson's Index (Simpson, 1949) based on probability theory and it defines the probability that a second individual drawn from an infinitely large population will be the same species as the first individual drawn from the same population (Southwood, 1978; Magurran, 1988).

The higher the score, the greater the equitability (Southwood, 1978). May (1975) showed that this index was strongly influenced by the number of species caught and by the underlying distribution.

A mathematically more simple index of dominance is the Berger-Parker index (Berger and Parker, 1970) which measures the amount of total species abundance which is accounted for by the most abundant species. Its inverse has been used to indicate increasing equitability with an increase in the value of the index. It is independent of S (species richness) but influenced by sample size (Magurran, 1988). May (1975) considered that it was one of the most satisfactory diversity measures available because it was easy to calculate and interpret. A related measure is the Two Dominant Species Index, which was more appropriate for the present study. The formula is written

$$d = \frac{N_{max} + N_{sec}}{N}$$

where N_{max} and N_{sec} represent the number of individuals in the most abundant and second most abundant species respectively, and N is the total number of individuals in the sample.

" The general message from the review of alpha-diversity measurements was that the ecological insights gained were by no means proportional to the mathematical sophistication and complexity of the methods." (Southwood, 1978; page 430)

Beta-diversity

Beta diversity is concerned with the change in species diversity from habitat to habitat and the comparison of quantitative and qualitative composition of different habitats or communities (Southwood, 1978). Clifford and Stephenson (1975) make the point that habitat definition is arbitrary because the difficulty in setting habitat boundaries by human vision is considerable and there is the ever present risk that the boundaries of the species will not fit a predetermined pattern.

Many of the methods for measuring beta-diversity have been developed and extensively used in plant ecology (Whittaker, 1972). Beta diversity is a measure of how similar or dissimilar a range of habitats or samples are in term of the variety and abundances of species present. The fewer species or characteristics that the communities share, the higher the beta diversity will be. Two simple beta diversity measures are Whittaker's measure and Cody's measure (see Magurran (1988) for formulae), and they both require presence/absence (binary) data and measure the number of species gained and lost along an environmental gradient.

An alternative approach is to investigate the degree of association or similarity of sites or samples using standard classification and ordination techniques (Magurran, 1988). The easiest way to measure the beta diversity between pairs of sites is to use similarity coefficients. A vast range of similarity and dissimilarity coefficients exist (Clifford and Stephenson, 1975) and the major problem of handling large data sets for comparing samples is the choice of the appropriate measure and the subsequent display of the information (Bullock, 1971). Association indices can be divided into those which are qualitative and measure binary data using presence or absence of a character or species, and quantitative indices which consider the species along with their abundances. Both indices are designed to equal unity (one) in the case of complete similarity and zero if the samples have no species in common.

Two commonly used qualitative indices are Jaccards Index and Sorensons Index (Magurran, 1988). These coefficients are simple to calculate and interpret but have been criticised in that all species are considered equally regardless of their abundance (Magurran, 1988), and tend therefore to place too much emphasis on rare species whose capture is more related to chance and sampling effort (Southwood, 1978). This consideration has led to similarity measures based on quantitative data (Southwood, 1978).

Many of the quantitative indices differ in their respect to the weight given to species abundance. Ludwig and Reynolds (1988) recognised three groups of dissimilarity measures: 1) The E-Group distances (the Euclidean coefficients); 2) the BC-Group (Bray-Curtis dissimilarity index) and 3) the RE-Group (the Relative

Euclidean distance measures).

The Euclidean measures are based on the principle that samples may be regarded as the culmination of information from a set of species, and these samples can be regarded as points in a species hyperspace (Bullock, 1971). Euclidean distance measures have been criticised on the grounds that they emphasise the larger differences in abundance between samples, and give considerable weighting to the abundant species. For this reason, Bullock (1971) and Clifford and Stephenson (1975) recommended transformation and or standardisation of the data before applying these indices. In spite of the popularity of these measures, Wolda (1981) and Ludwig and Reynold (1988) did not recommend their use. The RE-Group contains measures that are expressed as standardized or relative distance and performance was considered satisfactory over a wide range of data sets (Ludwig and Reynolds, 1988).

The Bray-Curtis group is represented by a single measure and is popular among ecologists (Ludwig and Reynolds, 1988). Wolda (1981) found this index could be influenced by sample size and so recommended instead the use of Morisita-Horn index (Horn, 1966). Two disadvantages of the latter coefficient are that it is highly sensitive to the abundance of the most abundant species, and that the actual value of the expected maximum was 'about one' (Wolda, 1981). This was considered a minor fault and the uncertaintities in not having a fixed upper limit were outweighed by the problems of correcting other indices for the effects of sample size. NESS (Normalised Expected Species Shared index) is a modified version of the Morisita index and it has also been used in ecological studies (e.g Didham, 1992). It is computationally complex and therefore less attractive than the Bray-Curtis measure. Nevertheless, the Bray Curtis coefficient has been highly recommended by Beals (1984) and has been applied to a wide range of ecological studies, especially in plant ecology (Goldsmith and Harrison, 1976) and was used in the present study for these reasons.

When there is a large number of sites in the investigation (more than 20), a good representation of beta-diversity can be obtained through cluster analysis and ordination (Magurran, 1988). These techniques give visual representations of habitat

or site similarity based on similarity measures and likeness of species assemblages. Some authors (e.g. Samways, 1984; Didham, 1992; Hutcheson, 1990) advocate that ordination and classification techniques describe community structure better and produce ecologically more-realistic groupings within the data than simple measures of diversity.

Cluster analysis is a classification technique for placing similar entities into groups or 'clusters'. These clusters are arranged into a tree like structure called a dendrogram. Numerous association functions ((dis)similarity indices described above) can be used to group the sample units (usually sites) and the resulting dendrogram is affected by different distance measures (Jongman *et al.*, 1987).

Cluster analysis can be either hierarchical or non-hierarchical (Jongman *et al.*, 1987). Non hierarchical clustering allows the scientist to choose sites which are thought to represent the extreme ends of the ecological continuum, and all other sites are arranged between these fixed end points. These methods are not discussed here.

Hierarchical techniques can be agglomerative or divisive. Divisive methods begin with all entities in a single group and split off subsets according to major differences between the sample units. Differences can be on a monothetic level (that is they are based on presence or absence of one characteristic or species) or polythetic (differences are based on several species or a combination of characteristics) (Ludwig and Reynolds, 1988). Two common divisive methods are Association analysis, and Two Way INdicator SPecies ANalysis (Twinspan). The latter has been popular in invertebrate studies (e.g. Hutcheson, 1990; Didham, 1992) and provides an indicator species alongside each branch of the dendrogram on which the site separations were based and a two-way table with the species arranged in a site by species matrix (Jongman *et al.*, 1987).

Agglomerative cluster analysis can be carried out with either qualitative or quantitative data, but in many cases the results are virtually identical (Magurran, 1988). Agglomerative methods of cluster analysis start with a matrix of similarity measures between each pair of sites. The two most similar sites are fused to form a

single cluster, and the process is repeated until all the sites are combined by a single branch of the dendrogram (Ludwig and Reynolds, 1988). There are various strategies of clustering 'like' groups and depending on the weighting (relative importance of the distance between groups) the resulting cluster formation will vary (Clifford and Stephenson, 1975).

Methods of clustering include Single Linkage or nearest neighbour clustering, Complete Linkage or furthest neighbour clustering, Centroid clustering and Average Linkage clustering (see Jongman *et al.*, (1987) for a more detailed explanation). For the latter measure, the between group (dis)similarity is defined as the average (dis)similarity between all possible pairs of members (one from each group). For any sample pair, the lowest dissimilarity (or highest similarity) is required to join them in the dendrogram. This method (also known as Unweighted Pair Group Method, UPGMA) is intermediate between complete and single linkage clustering techniques (Jongman *et al.*, 1987) and Clifford and Stephenson (1975) considered it more useful than the former measures.

The level of the horizontal linkage lines in the dendrogram rather than the order of the grouping is the significant part of this diagram (Southwood, 1978). Dendrograms and species by site tables cannot be used for presentation of data in more than one dimension (Jongman *et al.*, 1987). Homogenous communities are not amenable to classification. A continuous structure in the data set will almost always be obscured by a cluster analysis as it is arbitrarily partitioned into a discontinuous system of types or classes (Jongman *et al.*, 1987).

Ordination is a collective term for multivariate techniques which arrange sites along axes on the basis of data on species composition (ter Braak, 1987). Ordination techniques can be used to investigate the separation of sites according to ecological gradients (Magurran, 1988). They do not give a direct measure of beta-diversity *per se* but may be used to infer that there are a number of different communities present (Magurran, 1988).

Ordination is based on the concept of a continuum in which individual

samples are related to each other in a mathematical way which reflects their species composition and abundances. Coefficients of distance (dissimilarity) for each sample unit (site) can be plotted on an ordination diagram (Southwood, 1978). Sample points can be visualised and located in a hyperspace with their positions relative to one another being determined by their faunistic (or floristic) composition (Bullock, 1971). In ordination, the object is to sort samples so that they can be located with reference to a number of axes which are uncorrelated such that the maximum variation in the data is spread along the first axis, and successively smaller amounts of variation can be accounted for by the resulting spread in the second and subsequent axes (Bullock, 1971). Sites that are similar in species composition and species abundance occur close together on the ordination diagram, and dissimilar sites are far apart.

The total number of axes explaining the variation in the data is equal to the number of species present in the ordination. In most community studies, the first few axes extract the bulk of the original total variance, accounting for 40 to 90 percent of the variation within the data set (Gauch, 1982). In noisy data sets, the amount of variation explained is less, indicating that there is no single factor limiting or defining the species (and site) distribution.

Ordinations can be of two types: direct or indirect (ter braak, 1987). Direct methods position the sample units (sites) along measured environmental gradients which were selected by the researcher beforehand as the basis of the study. Indirect ordination (and cluster analysis) are the only available techniques when one has no measured environmental data, or the researcher is exploring the structure and the relationship of the sites to one another (Jongman *et al.*, 1988). Indirect ordination techniques arrange the sites within the axes based on their similarities and species composition and abundance. The factors separating the sites are unknown (theoretical), and the researcher makes inferences about the causative factors influencing site separation (Jongman *et al.*, 1987).

Popular direct ordination techniques include Principal Component Analysis (PCA), Correspondence Analysis (CA or RA, reciprocal Averaging) and techniques related to CA such as Detrended Correspondence Analysis (DCA). This latter

technique corrects for the arch effect found in PCA and CA, and compression of the first axis in RA (see Gauch, (1982) for explanation). Non metric multidimensional scaling (NMS) is an entire group of ordinations which replace the actual distance values between samples with their rank value, and is therefore a non-parametric equivalent of the PCA technique. It is computationally more complex however and these techniques are not used much in biology (Jongman *et al.*, 1987).

Direct techniques (or canonical ordination) allow the researcher to include a data set of environmental variables along with the ordination of the species data, and are designed to detect the patterns of variation in the species data that are 'best' explained by patterns in the environmental data (Jongman *et al.*, 1987). The species data are constrained to be linear combinations of the environmental variables, and the ordination diagram obtained has therefore a known environmental basis (ter Braak, 1988). Environmental gradients are then represented by coordinates coding for the head of an arrow diverging from the origin of a species ordination diagram. These arrows represent the direction of influence for each environmental variable. Environmental variables with long arrows more strongly influence the separation of species than those with short arrows. The length of the arrow is proportional to the amount of variance of the data explained by that particular gradient (ter Braak, 1988). Sites at opposite ends of the arrows can best be separated using that environmental variable (Fielding and Brusven, 1993).

Direct ordination techniques include Redundancy Analysis (RDA), Canonical Correspondence Analysis (CCA) and Detrended Canonical Correspondence Analysis (DCCA). DCCA is an efficient ordination technique when species have a bell shaped (Gaussian) response curve or surface with resect to environmental gradients and is therefore more appropriate for analysing data on community composition and environmental variables. The method would not work if a large number of species was distributed in a more complex way (Ter Braak, 1986). Canonical correspondence analysis has the advantage over other techniques in that it focuses on the relations between species and environmental variables and so provides an automated interpretation of the ordination axes (Ter Braak, 1986). DCA and DCCA were the ordination techniques used in the present study.

Appendix 4

Invertebrate species list 1 (in numerical order).
Numbers are species identification codes.

1 Mites
6 *Nysius huttoni*
7 Phoridae sp 1
8 Phoridae sp 2
10 Scydmaenidae
11 *Monomorium antarcticum*
16 Gnaphosidae sp 1
17 *Nesomymar* sp 1
18 *Trichopria* sp 1
19 *Telenomus* sp
20 *Odiaglyptus biformis*
21 *Phalangium opilio*
24 *Sigaus minutus*
25 *Hemiandrus* new sp
26 Entomobryoidea Collembola
27 Poduroidea Collembola
28 Neelipleona/Symphypleona Collembola
29 Aphididae sp 1
32 *Otiorhynchus ovatus*
33 Trichoptera
35 Toxopidae sp 1
36 Aphididae sp 2
37 Sciaridae
39 Noctuidae
41 *Costelytra zelandica*
42 *Actenonyx bembidioides*
43 Gnaphosidae sp 2
47 Chilopoda
54 *Phytomyza* sp
55 *Gryon* sp
59 Cicadellidae sp 1
62 Pselaphidae sp 1
63 Gryllidae
64 *Bembidion* sp 1
65 *Bembidion* sp 2
66 *Neocicindela feredayi*
67 *Rhopus anceps*
68 Elateridae sp 2
70 *Melanopthalma* sp
72 unid. bug (juv)
74 *Trichogramma* sp
76 Linyphiidae sp 1
77 Chironomidae
79 *Spilomicrus* sp 1
80 *Baeus* sp 1
83 Dictynidae sp 1
84 *Anisodactylus binotatus*
85 Lepidoptera larvae
88 Margarodidae
89 Coleoptera larvae
91 *Limosina* sp
92 *Hyperodes bonariensis*
93 Byrrhidae sp 1
94 Thysanoptera
95 *Notogonum feredayi*
98 Diptera larvae
100 Agelenidae sp 1
101 Tipulidae
102 *Trimorus* sp 1
104 Staphylinidae sp 2
105 *Neobaeus ?novazealandensi*
106 Unid. weevil. sp 1
107 *Smittia* sp
109 Trichopria sp 2
111 *Conocephalus* sp
112 Theridiidae sp 1
113 Muscidae sp 1
116 *Scatella* new sp
117 Oedomeridae
120 Colydiidae sp 1
122 *Austrosimulium* sp
123 Staphylinidae sp 3
124 *Coccinella undecimpunctata*
125 Byrrhidae sp 2
126 *Forficula auricularia*
127 Snail
129 Earthworm
130 Staphylinidae sp 4
133 *Saldula* sp
135 Cecidomyiidae
136 Diaspididae sp
139 Hahniidae sp 1
140 Gnaphosidae sp 3
143 Moth 2
144 Moth 3
145 *Telenomus* spp
149 *Nematopus* sp
150 *Priocnemis nitidiventris*
151 Dictynidae sp 2

152 Eulophidae sp 1
153 Cercopidae sp 1
154 *Hypharpax* sp 1
155 *Lasioglossum?* sp
156 Linyphiidae sp 2
157 Cercopidae sp 2
158 Cicadellidae sp 2
159 Pteromalidae sp 1
160 *Psilopa metallica*
161 *Cardiopria* sp
162 Psocidae?
163 Braconidae spp
164 *Stethynium?* sp
168 Phoridae sp 3
170 Mycetophagidae
171 *Trichopria* sp 3
173 Moth 4
174 *Trichomalopsis* sp 1
175 *Porcellio scaber*
176 Agelenidae sp 2
177 *Hemideina maori*
178 *Baeus* sp 2
179 Bethylidae sp 1
180 Phoridae sp 4
181 Cicadellidae sp 3
182 Eulophidae sp 2
183 *Ostenia?* sp
184 Eucinetidae sp
185 Lygaeidae sp 2
186 Tachinidae sp 1
188 Agromyzidae sp 1
189 *Scatella ?nelsoni*
190 Pselaphidae sp 2
191 Dictynidae sp 3
192 *Botanophila* sp
193 Moth 5
194 False scorpion
195 *Zizina otis oxleyi*
196 Ephemeroptera
197 Ceraphronidae sp 1
198 Dolichopodidae sp 1
199 Tachinidae sp 2
202 Gnaphosidae sp 4
203 Phoridae sp 5
204 Cicadellidae sp 4
205 *Monomorium* sp
206 *Phaulacridium marginale*
208 *Odontomyia* sp
209 *Cerodontha australis*
210 *Anabarrhynchus* sp
211 Cicadidae sp
212 Linyphiidae sp 3
213 Millipede
214 *Podagritus albiceps*
215 *Demetrida dieffenbachi*
217 Culicidiae
218 *Basalys* sp
219 Slug
220 Bethylidae sp 2
221 Stiphidiidae
223 *Anacharis zealandica*
224 Hahniidae sp 2
226 Neuroptera
228 Mycetophilidae spp
229 Moth 7
230 Salticidae sp 1
231 Mycetophilidae sp 1
232 Mycetophilidae sp 2
233 Encyrtidae sp
236 Cicadellidae sp 5
237 Flatworm
238 *Mecyclothorax rotundicollis*
239 Lygaeidae sp 4
240 Phoridae sp 6
241 Staphylinidae sp 7
242 *Notogonum submetallicum*
243 *Metaglymma tersatum*
244 Agromyzidae sp 2
245 *Neoscatella vittithorax*
246 *Scatella nubeculosa*
248 *Baeus* sp 3
249 *Neolimnia ?minuta*
250 Ichneumonidae sp 2
251 Ichneumonidae sp 3
252 *Pontania proxima*
253 *Trichopria* sp 2
254 Muscidae sp 2
255 Unid. beetle. sp 1
256 *Erigone wiltoni*
257 *Coccinella leonina*
258 Cicadellidae sp 6
261 *Apis mellifera*
262 Salticidae sp 2
263 Pteromalidae sp 1
265 Ichneumonidae sp 4
266 Mymaridae sp 2
267 Staphylinidae sp 8
268 *Ascogaster tekapoense*
269 Asilidae sp
270 Megaspilidae sp 1
271 *Mirotelenomus* sp
272 Pipunculidae
273 Muscidae sp 3
275 Platygastridae sp
277 Anthribidae sp
278 Ceraphronidae sp 4
279 Chloropidae sp 1
280 *Megastigmus aculeatus*
282 Ceraphronidae spp
283 *Archeocrypticus topali*
284 *Zealandonota?* sp
285 *Spilomicrus* sp 2
286 Chloropidae sp 2
288 Ephydridae sp
289 *Otiorhynchus sulcatus*
290 *Sitona discoideus*
291 Unid. weevil. sp 2
292 Dolichopodidae sp 2
293 Dictynidae sp 3
294 Lygaeidae sp 3
295 *Macroneura vesicularis*
296 *Bothrideres* sp 1
297 Muscidae sp 4
299 *Bothrideres* sp 2
300 Dictynidae sp 4
301 Unid. Spid. sp 1
302 Hahniidae sp 3
303 Agromyzidae sp 3
304 Hydrophilidae sp 1
305 *Hypharpax* sp 2
306 Muscidae sp 5
307 *Microvelia* sp
308 Muscidae sp 6
309 Unid. weevil. sp 3
310 Hydrophilidae sp 2
311 *Scelio* sp
312 Alysiinae
313 Braconidae sp 1
314 *Kleidotoma* sp 1
315 Eulophidae sp 3
316 *?Apanteles* sp
317 *Trimorus* sp 2
318 *Spalangia* sp
319 Miridae sp

Class or Order	Suborder	Division	Superfamily	Family	Tribe	Taxa identification	Taxa id #
Acari						Mites	1
Annelida						Earthworm	129
Arachnida						Phalangium opilio	21
Araneae						Agelenidae sp 1	100
Araneae						Agelenidae sp 2	176
Araneae						Dictynidae sp 1	151
Araneae						Dictynidae sp 2	83
Araneae						Dictynidae sp 3	191
Araneae						Dictynidae sp 3	293
Araneae						Dictynidae sp 4	300
Araneae						Gnaphosidae sp 1	16
Araneae						Gnaphosidae sp 2	43
Araneae						Gnaphosidae sp 3	140
Araneae						Gnaphosidae sp 4	202
Araneae						Hahniidae sp 1	139
Araneae						Hahniidae sp 2	224
Araneae						Hahniidae sp 3	302
Araneae						Linyphiidae sp 1	76
Araneae						Linyphiidae sp 2	156
Araneae						Linyphiidae sp 3	212
Araneae						Erigone wiltoni	256
Araneae						Salticidae sp 1	230
Araneae						Salticidae sp 2	262
Araneae						Stiphidiidae	221
Araneae						Theridiidae sp 1	112
Araneae						Toxopidae sp 2	35
Araneae						Unid. Spid. sp 1	301
Chilopoda						Chilopoda	47
Coleoptera	Adephaga		Caraboidea	Carabidae		Actenonyx bembidioides	42
Coleoptera	Adephaga		Caraboidea	Carabidae		Bembidion sp 1	64
Coleoptera	Adephaga		Caraboidea	Carabidae		Bembidion sp 2	65
Coleoptera	Adephaga		Caraboidea	Carabidae		Demetrida dieffenbachi	215
Coleoptera	Adephaga		Caraboidea	Carabidae		Metaglymma tersatum	243

Coleoptera	Adephaga	Caraboidea	Carabidae	Neocicindela feredayi	66
Coleoptera	Adephaga	Caraboidea	Carabidae	Notogonum feredayi	95
Coleoptera	Adephaga	Caraboidea	Carabidae	Notogonum submetallicum	242
Coleoptera	Polyphaga	Byrrhoidea	Byrrhidae	Byrrhidae sp 1	93
Coleoptera	Polyphaga	Byrrhoidea	Byrrhidae	Byrrhidae sp 2	125
Coleoptera	Polyphaga	Carabiodea	Carabidae	Anisodactylus binotatus	84
Coleoptera	Polyphaga	Caraboidea	Carabidae	Hypharpax sp 1	154
Coleoptera	Polyphaga	Caraboidea	Carabidae	Hypharpax sp 2	305
Coleoptera	Polyphaga	Caraboidea	Carabidae	Mecyclothorax rotundicollis	238
Coleoptera	Polyphaga	Cucujoidea	Bothrideridae	Bothrideres sp 1	296
Coleoptera	Polyphaga	Cucujoidea	Bothrideridae	Bothrideres sp 2	299
Coleoptera	Polyphaga	Cucujoidea	Coccinellidae	Coccinella leonina	257
Coleoptera	Polyphaga	Cucujoidea	Coccinellidae	Coccinella undecimpunctata	124
Coleoptera	Polyphaga	Cucujoidea	Lathridiidae	Melanopthalma sp	70
Coleoptera	Polyphaga	Curculionoidea	Anthribidae	Anthribidae sp	277
Coleoptera	Polyphaga	Curculionoidea	Curculionidae	Otiorhynchus ovatus	32
Coleoptera	Polyphaga	Curculionoidea	Curculionidae	Otiorhynchus sulcatus	289
Coleoptera	Polyphaga	Curculionoidea	Curculionidae	Sitona discoideus	290
Coleoptera	Polyphaga	Curculionoidea	Curculionidae	Unid. weevil. sp 2	291
Coleoptera	Polyphaga	Curculionoidea	Curculionidae	Unid. weevil. sp 3	309
Coleoptera	Polyphaga	Curculionoidea		Unid. weevil. sp 1	106
Coleoptera	Polyphaga	Curculionoidea		Hyperodes bonariensis	92
Coleoptera	Polyphaga	Elateroidea	Elateridae	Elateridae sp 2	68
Coleoptera	Polyphaga	Eucinetoidea	Eucinetidae	Eucinetidae sp	184
Coleoptera	Polyphaga	Hydrophiloidea	Hydrophilidae	Hydrophilidae sp 1	304
Coleoptera	Polyphaga	Hydrophiloidea	Hydrophilidae	Hydrophilidae sp 2	310
Coleoptera	Polyphaga	Scaraboidea	Scarabaeidae	Costelytra zelandica	41
Coleoptera	Polyphaga	Staphylinoidea	Pselaphidae	Pselaphidae sp 1	62
Coleoptera	Polyphaga	Staphylinoidea	Pselaphidae	Pselaphidae sp 2	190
Coleoptera	Polyphaga	Staphylinoidea	Scydmaenidae	Scydmaenidae sp	10
Coleoptera	Polyphaga	Staphylinoidea	Staphylinidae	Staphylinidae sp 2	104
Coleoptera	Polyphaga	Staphylinoidea	Staphylinidae	Staphylinidae sp 3	123
Coleoptera	Polyphaga	Staphylinoidea	Staphylinidae	Staphylinidae sp 4	130
Coleoptera	Polyphaga	Staphylinoidea	Staphylinidae	Staphylinidae sp 7	241
Coleoptera	Polyphaga	Staphylinoidea	Staphylinidae	Staphylinidae sp 8	267
Coleoptera	Polyphaga	Tenebrionoidea	Archaeocryptidae	Archeocrypticus topali	283

Coleoptera	Polyphaga		Tenebrionoidea	Colydiidae	Colydiidae sp 1	120
Coleoptera	Polyphaga		Tenebrionoidea	Mycetophagidae	Mycetophagidae sp	170
Coleoptera	Polyphaga		Tenebrionoidea	Oedomeridae	Oedomeridae sp	117
Coleoptera	Polyphaga				Coleoptera larvae	89
Coleoptera	Polyphaga				Unid. beetle. sp 1	255
Collembola					Collembola sp 1	26
Collembola					Collembola sp 2	27
Collembola					Collembola sp 3	28
Dermaptera					Forficula auricularia	126
Diptera	Brachycera	Cyclorrhapha	Phoroidea	Phoridae	Phoridae sp 1	7
Diptera	Brachycera	Cyclorrhapha	Phoroidea	Phoridae	Phoridae sp 2	8
Diptera	Brachycera	Cyclorrhapha	Opomyzoidea	Agromyzidae	Phytomyza sp	54
Diptera	Brachycera	Cyclorrhapha	Heleomyzoidea	Sphaeroceridae	Limosina sp	91
Diptera	Brachycera	Cyclorrhapha	Muscoidea	Muscidae	Muscidae sp 1	113
Diptera	Brachycera	Cyclorrhapha	Ephydroidea	Ephydridae	Scatella new sp	116
Diptera	Brachycera	Cyclorrhapha	Ephydroidea	Ephydridae	Psilopa metallica	160
Diptera	Brachycera	Cyclorrhapha	Phoroidea	Phoridae	Phoridae sp 3	168
Diptera	Brachycera	Cyclorrhapha	Phoroidea	Phoridae	Phoridae sp 4	180
Diptera	Brachycera	Cyclorrhapha	Muscoidea	Tachinidae	Tachinidae sp 1	186
Diptera	Brachycera	Cyclorrhapha	Opomyzoidea	Agromyzidae	Agromyzidae sp 1	188
Diptera	Brachycera	Cyclorrhapha	Ephydroidea	Ephydridae	Scatella ?nelsoni	189
Diptera	Brachycera	Cyclorrhapha	Muscoidea	Anthomyiidae	Botanophila sp	192
Diptera	Brachycera	Cyclorrhapha	Muscoidea	Tachinidae	Tachinidae sp 2	199
Diptera	Brachycera	Cyclorrhapha	Phoroidea	Phoridae	Phoridae sp 5	203
Diptera	Brachycera	Cyclorrhapha	Opomyzoidea	Agromyzidae	Cerodontha australis	209
Diptera	Brachycera	Cyclorrhapha	Phoroidea	Phoridae	Phoridae sp 6	240
Diptera	Brachycera	Cyclorrhapha	Opomyzoidea	Agromyzidae	Agromyzidae sp 2	244
Diptera	Brachycera	Cyclorrhapha	Ephydroidea	Ephydridae	Neoscatella vittithorax	245
Diptera	Brachycera	Cyclorrhapha	Ephydroidea	Ephydridae	Scatella nubeculosa	246
Diptera	Brachycera	Cyclorrhapha	Sciomyzoidea	Sciomyzidae	Neolimnia ?minuta	249
Diptera	Brachycera	Cyclorrhapha	Muscoidea	Muscidae	Muscidae sp 2	254
Diptera	Brachycera	Cyclorrhapha	Asiloidea	Asilidae	Asilidae sp	269

Diptera	Brachycera	Cyclorrhapha	Syrphoidea	Pipunculidae	Pipunculidae sp	272
Diptera	Brachycera	Cyclorrhapha	Muscoidea	Muscidae	Muscidae sp 3	273
Diptera	Brachycera	Cyclorrhapha	Chloropoidea	Chloropidae	Chloropidae sp 1	279
Diptera	Brachycera	Cyclorrhapha	Chloropoidea	Chloropidae	Chloropidae sp 2	286
Diptera	Brachycera	Cyclorrhapha	Ephydroidea	Ephydridae	Ephydridae sp	288
Diptera	Brachycera	Cyclorrhapha	Empidoidea	Dolichopodidae	Dolichopodidae sp 2	292
Diptera	Brachycera	Cyclorrhapha	Muscoidea	Muscidae	Muscidae sp 4	297
Diptera	Brachycera	Cyclorrhapha	Opomyzoidea	Agromyzidae	Agromyzidae sp 3	303
Diptera	Brachycera	Cyclorrhapha	Muscoidea	Muscidae	Muscidae sp 5	306
Diptera	Brachycera	Cyclorrhapha	Muscoidea	Muscidae	Muscidae sp 6	308
Diptera	Brachycera	Orthorrhapha	Empidoidea	Dolichopodidae	Nematopus sp	149
Diptera	Brachycera	Orthorrhapha	Empidoidea	Dolichopodidae	Ostenia? sp	183
Diptera	Brachycera	Orthorrhapha	Empidoidea	Dolichopodidae	Dolichopodidae sp 1	198
Diptera	Brachycera	Orthorrhapha	Tabanoidea	Stratiomyidae	Odontomyia sp	208
Diptera	Brachycera	Orthorrhapha	Asiloidea	Therevidae	Anabarrhynchus sp	210
Diptera	Nematocera	Bibionomorpha		Chironomidae	Sciariidae sp	37
Diptera	Nematocera	Bibionomorpha		Cecidomyidae	Cecidomyidae sp	135
Diptera	Nematocera	Bibionomorpha	Chironomoidea	Mycetophilidae	Mycetophilidae spp	228
Diptera	Nematocera	Bibionomorpha	Chironomoidea	Mycetophilidae	Mycetophilidae sp 1	231
Diptera	Nematocera	Bibionomorpha	Chironomoidea	Mycetophilidae	Mycetophilidae sp 2	232
Diptera	Nematocera	Culicomorpha	Chironomoidea	Chironomidae	Chironomidae sp	77
Diptera	Nematocera	Culicomorpha	Chironomoidea	Chironomidae	Smittia sp	107
Diptera	Nematocera	Culicomorpha	Chironomoidea	Simuliidae	Austrosimulium sp	122
Diptera	Nematocera	Culicomorpha	Culicoidea	Culicidiae	Culicidiae sp	217
Diptera	Nematocera	Tipulomorpha	Tipuloidea	Tipulidae	Limonia sp	101
Diptera					Diptera larvae	98
Ephemeroptera					Ephemeroptera	196
Hemiptera	Auchenorrhyncha		Cercopoidea	Cercopidae	Cercopidae sp 1	153
Hemiptera	Auchenorrhyncha		Cercopoidea	Cercopidae	Cercopidae sp 2	157
Hemiptera	Auchenorrhyncha		Cicadelloidea	Cicadellidae	Cicadellidae sp 1	59
Hemiptera	Auchenorrhyncha		Cicadelloidea	Cicadellidae	Cicadellidae sp 2	158
Hemiptera	Auchenorrhyncha		Cicadelloidea	Cicadellidae	Cicadellidae sp 3	181
Hemiptera	Auchenorrhyncha		Cicadelloidea	Cicadellidae	Cicadellidae sp 4	204
Hemiptera	Auchenorrhyncha		Cicadelloidea	Cicadellidae	Cicadellidae sp 5	236

Hemiptera	Auchenorrhyncha	Cicadelloidea	Cicadellidae		Cicadellidae sp 6	258
Hemiptera	Auchenorrhyncha	Cicadoidea	Cicadidae		Cicadidae sp	211
Hemiptera	Heteroptera	Gerroidea	Veliidae		Microvelia sp	307
Hemiptera	Heteroptera	Lygaeoidea	Lygaeidae		Nysius huttoni	6
Hemiptera	Heteroptera	Lygaeoidea	Lygaeidae		Lygaeidae sp 2	185
Hemiptera	Heteroptera	Lygaeoidea	Lygaeidae		Lygaeidae sp 4	239
Hemiptera	Heteroptera	Lygaeoidea	Lygaeidae		Lygaeidae sp 3	294
Hemiptera	Heteroptera	Miroidea	Miridae		Miridae sp	319
Hemiptera	Heteroptera	Saldoidea	Saldidae		Saldula sp	133
Hemiptera	Sternorhyncha	Aphidoidea	Aphididae		Aphididae sp 1	29
Hemiptera	Sternorhyncha	Aphidoidea	Aphididae		Aphididae sp 2	36
Hemiptera	Sternorhyncha	Coccoidea	Margarodidae		Margarodidae sp	88
Hemiptera	Sternorhyncha	Coccoidea	Diaspididae		Diaspididae sp	136
Hemiptera					unid. bug (juv)	72
Hymenoptera		Ceraphronoidea	Ceraphronidae		Ceraphronidae sp 1	197
Hymenoptera		Ceraphronoidea	Megaspilidae		Megaspilidae sp 1	270
Hymenoptera		Ceraphronoidea	Ceraphronidae		Ceraphronidae sp 4	278
Hymenoptera		Ceraphronoidea	Ceraphronidae		Ceraphronidae spp	282
Hymenoptera		Chacidoidea	Mymaridae		Mymaridae sp 1	266
Hymenoptera		Chalcidoidea	Pteromalidae	Spalangiinae	Spalangia sp	318
Hymenoptera		Chalcidoidea	Encyrtidae	Tetracneminae	Odiaglyptus biformis	20
Hymenoptera		Chalcidoidea	Encyrtidae	Tetracneminae	Rhopus anceps	67
Hymenoptera		Chalcidoidea	Mymaridae		Nesomymar sp 1	17
Hymenoptera		Chalcidoidea	Trichogrammatidae		Trichogramma sp	74
Hymenoptera		Chalcidoidea	Eulophidae		Eulophidae sp 1	152
Hymenoptera		Chalcidoidea	Pteromalidae		Pteromalidae sp 1	159
Hymenoptera		Chalcidoidea	Mymaridae		Stethynium? sp	164
Hymenoptera		Chalcidoidea	Pteromalidae		Trichomalopsis sp 1	174
Hymenoptera		Chalcidoidea	Eulophidae		Eulophidae sp 2	182
Hymenoptera		Chalcidoidea	Encyrtidae		Encyrtidae sp	233
Hymenoptera		Chalcidoidea	Pteromalidae		Pteromalidae sp 1	263
Hymenoptera		Chalcidoidea	Torymidae		Megastigmus aculeatus	280
Hymenoptera		Chalcidoidea	Eupelmidae		Macroneura vesicularis	295
Hymenoptera		Chalcidoidea	Eulophidae		Eulophidae sp 3	315
Hymenoptera		Chrysidoidea	Bethylidae		Bethylidae sp 1	179

Hymenoptera	Chrysidoidea			Bethylidae sp 2	220
Hymenoptera	Cynipoidea	Figitidae		Anacharis zealandica	223
Hymenoptera	Cynipoidea	Eucoilidae		Kleidotoma sp 1	314
Hymenoptera	Ichneumonoidea	Braconidae	Alysiinae	Alysiinae sp	312
Hymenoptera	Ichneumonoidea	Braconidae	Cheloninae	Ascogaster tekapoense	268
Hymenoptera	Ichneumonoidea	Braconidae	Microgastrinae	?Apanteles sp	316
Hymenoptera	Ichneumonoidea	Braconidae		Braconidae spp	163
Hymenoptera	Ichneumonoidea	Ichneumonidae		Ichneumonidae sp 1	250
Hymenoptera	Ichneumonoidea	Ichneumonidae		Ichneumonidae sp 2	251
Hymenoptera	Ichneumonoidea	Ichneumonidae		Ichneumonidae sp 3	265
Hymenoptera	Ichneumonoidea	Braconidae		Braconidae sp 1	313
Hymenoptera	Proctotrupoidea	Diapriidae	Diapriinae	Trichopria sp 1	18
Hymenoptera	Proctotrupoidea	Diapriidae	Diapriinae	Spilomicrus sp 1	79
Hymenoptera	Proctotrupoidea	Diapriidae	Diapriinae	Cardiopria sp	161
Hymenoptera	Proctotrupoidea	Diapriidae	Diapriinae	Trichopria sp 3	171
Hymenoptera	Proctotrupoidea	Diapriidae	Diapriinae	Basalys sp	218
Hymenoptera	Proctotrupoidea	Diapriidae	Diapriinae	Trichopria sp 4	253
Hymenoptera	Proctotrupoidea	Diapriidae	Diapriinae	Spilomicrus sp 2	285
Hymenoptera	Proctotrupoidea	Diapriidae		Trichopria sp 2	109
Hymenoptera	Scelionoidea	Scelionidae	Scelioninae	Gryon sp	55
Hymenoptera	Scelionoidea	Scelionidae	Scelioninae	Baeus sp 1	80
Hymenoptera	Scelionoidea	Scelionidae	Scelioninae	Neobaeus ?novazealandensi	105
Hymenoptera	Scelionoidea	Scelionidae	Scelioninae	Baeus sp 2	178
Hymenoptera	Scelionoidea	Scelionidae	Scelioninae	Baeus sp 3	248
Hymenoptera	Scelionoidea	Scelionidae	Scelioninae	Mirotelenomus sp	271
Hymenoptera	Scelionoidea	Scelionidae	Scelioninae	Scelio sp	311
Hymenoptera	Scelionoidea	Scelionidae	Teleasinae	Trimorus sp 1	102
Hymenoptera	Scelionoidea	Scelionidae	Teleasinae	Trimorus sp 2	317
Hymenoptera	Scelionoidea	Scelionidae	Telenominae	Telenomus sp	19
Hymenoptera	Scelionoidea	Scelionidae	Telenominae	Telenomus spp	145
Hymenoptera	Scelionoidea			Platygastridae sp	275
Hymenoptera	Scelionoidea	Platygastridae		Zealandonota? sp	284
Hymenoptera	Sphecoidea	Sphecidae	Crabroninae	Podagritus albiceps	214
Hymenoptera	Sphecoidea	Halictidae	Halictinae	Lasioglossum? sp	155
Hymenoptera	Tenthredinoidea	Tenthredinidae		Pontania proxima	252
Hymenoptera	Vespoidea	Pompilidae		Priocnemis nitidiventris	150

Hymenoptera	Formicidae	Monomorium antarcticum	11
Hymenoptera	Formicidae	Monomorium sp	205
Hymenoptera		Apis mellifera	261
Isopoda		Porcellio scaber	175
Lepidoptera		Lepidoptera larvae	85
Lepidoptera		Moth 2	143
Lepidoptera		Moth 3	144
Lepidoptera		Moth 4	173
Lepidoptera		Moth 5	193
Lepidoptera		Moth 7	229
Lepidoptera		Noctuidae	39
Lepidoptera		Zizina otis oxleyi	195
Mollusca		Snail	127
Mollusca		Slug	219
Myriopoda		Millipede	213
Neuroptera		Neuroptera	226
Orthoptera		Gryllidae	63
Orthoptera		Sigaus minutus	24
Orthoptera		Conocephalus sp	111
Orthoptera		Phaulacridium marginale	206
Orthoptera		Hemideina maori	177
Orthoptera		Hemiandrus new sp	25
Pseudoscorpiones		False scorpion	194
Psocoptera		Psocidae	162
Thysanoptera		Thysanoptera	94
Trichoptera		Trichoptera	33
Turbellaria		Flatworm	237

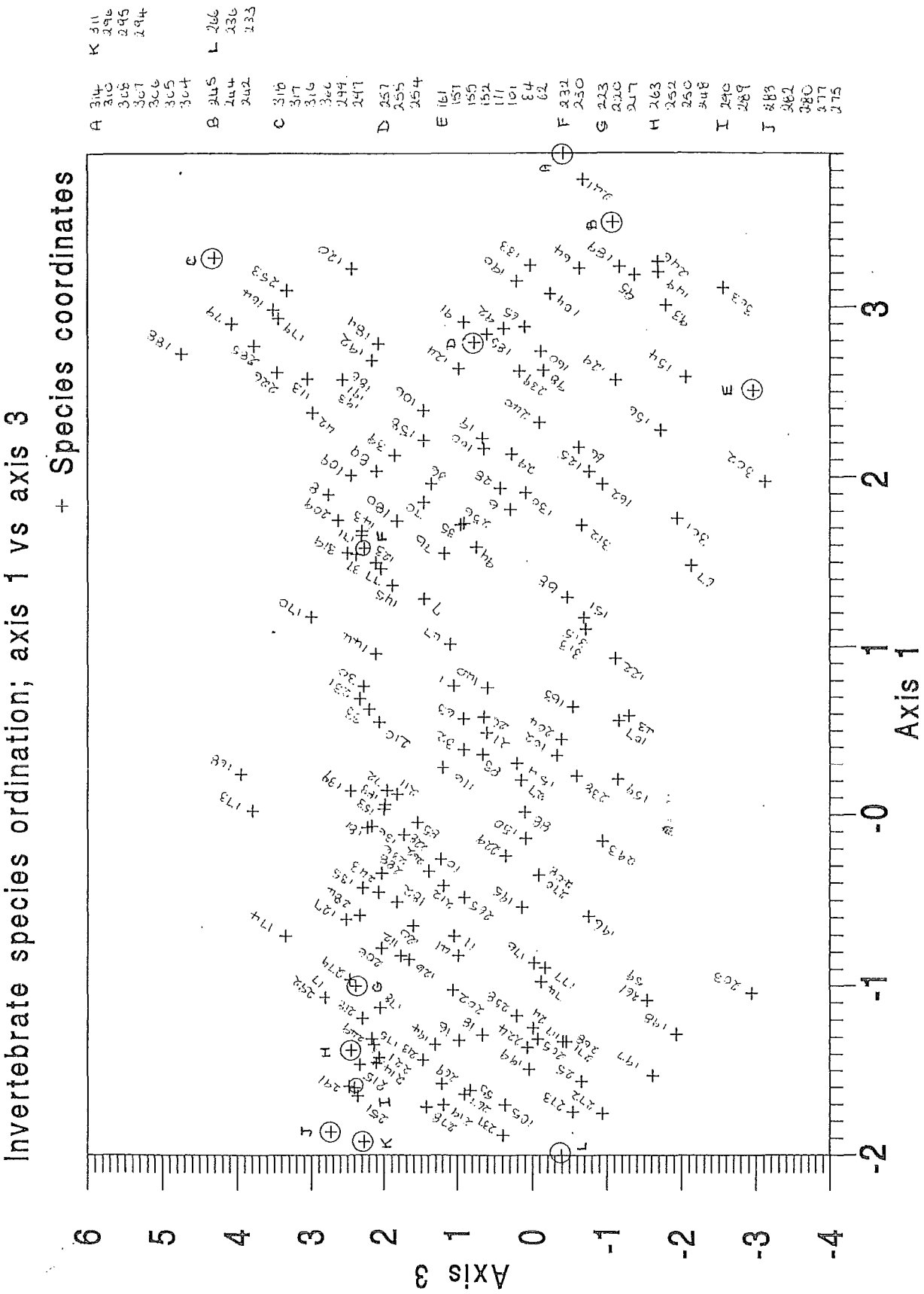
Appendix 5

Plant species list. Numbers are species identification codes.

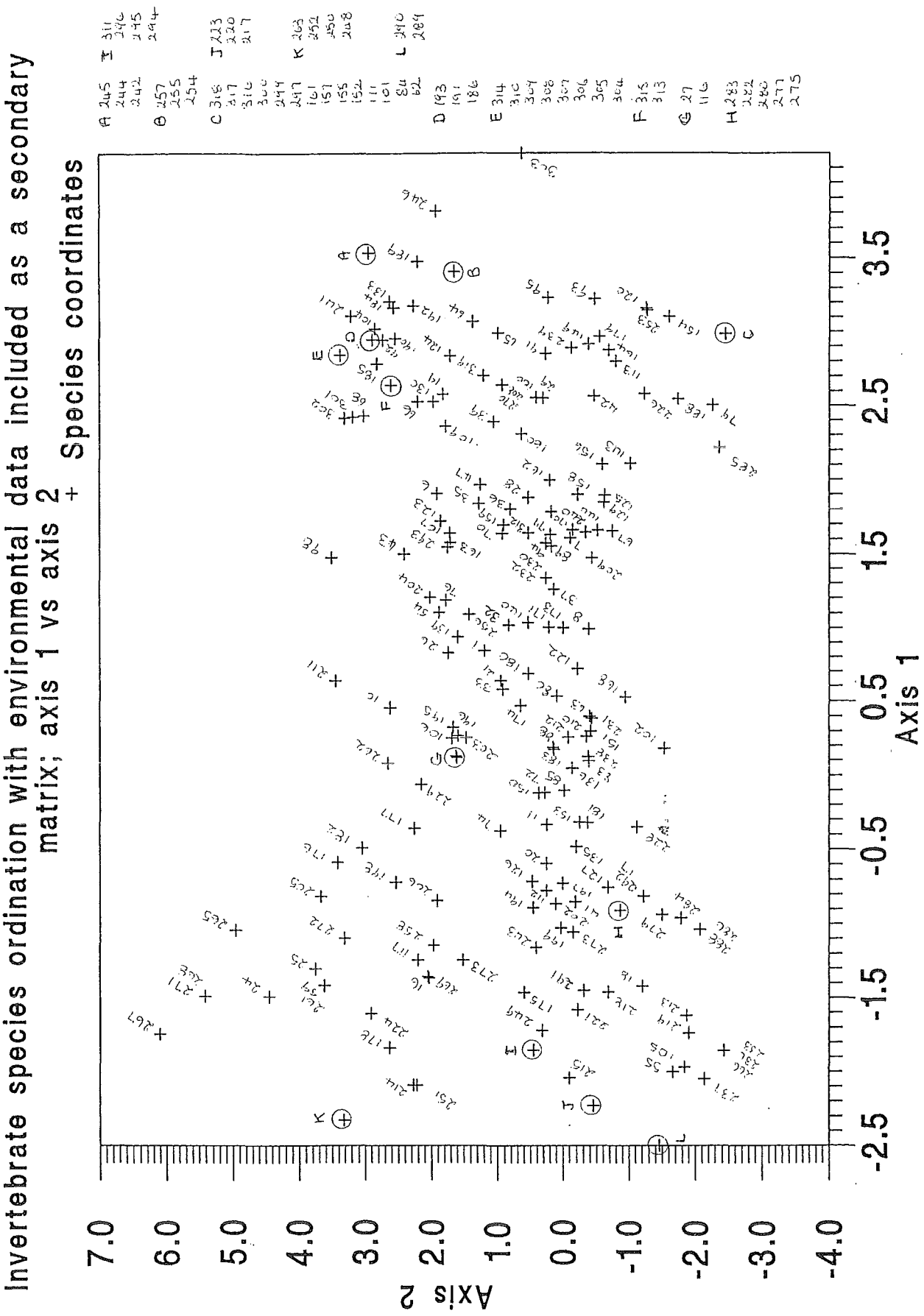
1 *Acaena* sp
2 *Agrostis capillaris*
3 *Aira caryophyllea*
5 *Anthoxanthum odoratum*
6 *Arenaria serpyllifolia*
7 *Barbarea* sp
9 *Cerastium semidecandrum*
10 *Cirsium vulgare*
12 *Crepis capillaris*
13 *Cuscuta epithymum*
15 *Discaria toumatou*
17 *Elymus rectisetus*
18 *Epilobium melanocaulon*
19 *Epilobium* cf *ciliatum*
20 *Erodium* sp
21 *Eschscholzia californica*
22 *Festuca ovina*
23 *Festuca rubra* var. *commutata*
24 *Gonocarpus aggregatus*
26 *Hieracium* spp (*praealtum* + *?aurantiacum*)
28 *Hieracium pilosella*
29 *Holcus mollis*
30 *Hypericum perforatum*
31 *Hypochoeris radicata*
32 *Juncus articulatus*
33 *Juncus* cf *tenuis*
34 *Linum catharticum*
35 *Lolium perenne*
36 *Lotus pedunculatus*
38 *Medicago lupulina*
39 *Melilotus officinalis*
40 *Mimulus guttatus*
41 *Muehlenbeckia axillaris*
42 *Myosotis laxa*
43 *Plantago lanceolata*
45 *Polygonum persicaria*
46 *Prunella vulgaris*
47 *Racomitrium lanuginosum*
48 *Raoulia* spp (*?australis, ?hookeri, ?haastii*)
50 *Raoulia* cf *tenuicaulis*
51 *Rosa rubiginosa*
52 *Rumex acetosella*
53 *Rumex* cf *crispus*
54 *Sagina procumbens*
55 *Salix fragilis*

56 *Sanguisorba minor*
57 *Sedum acre*
59 *Stellaria graminea*
61 *Taraxacum officinale*
62 *Trifolium arvense*
63 *Trifolium repens*
64 *Usnea*? lichen
65 *Verbascum thapsus*
67 *Vulpia* cf *myuros*
69 *Logfia minima*
71 *Bromus tectorum*
72 Black rock lichen
74 Teal rock lichen
76 Fawn scaly lichen
77 *Echium vulgare*
78 *Leucanthemum vulgare*
79 *Spergularia rubra*
80 *Leontodon taraxacoides*
81 *Poa* sp
82 *Epilobium rostratum*
83 *Dactylis glomerata*
84 *Cirsium arvense*
85 *Collomia cavanillesii*
86 *Galium* sp
87 *Cerastium fontanum*
88 *Agrostis stolonifera*
91 *Rhizocarpon*? rock lichen
97 Cup lichen
99 *Coriaria* sp
102 *Plantago major*
103 *Isolepis setacea*
104 *Rytidosperma maculatum*
105 *Carex* sp 1
110 *Oxalis* cf *corniculata*
111 *Dianthus armeria*
112 *Achillea millefolium*
113 *Trifolium pratense*
114 *Cytisus scoparius*
115 *Ranunculus repens*
116 *Salix* x *reichardtii*
117 *Trifolium dubium*
118 *Stellaria alsine*
119 *Epilobium microphyllum*
121 *Phleum pratense*
123 *Navarretia squarrosa*
125 *Veronica* cf *serpyllifolia*
126 *Eleocharis acuta*
128 *Rumex obtusifolius*
129 *Juncus effusus*

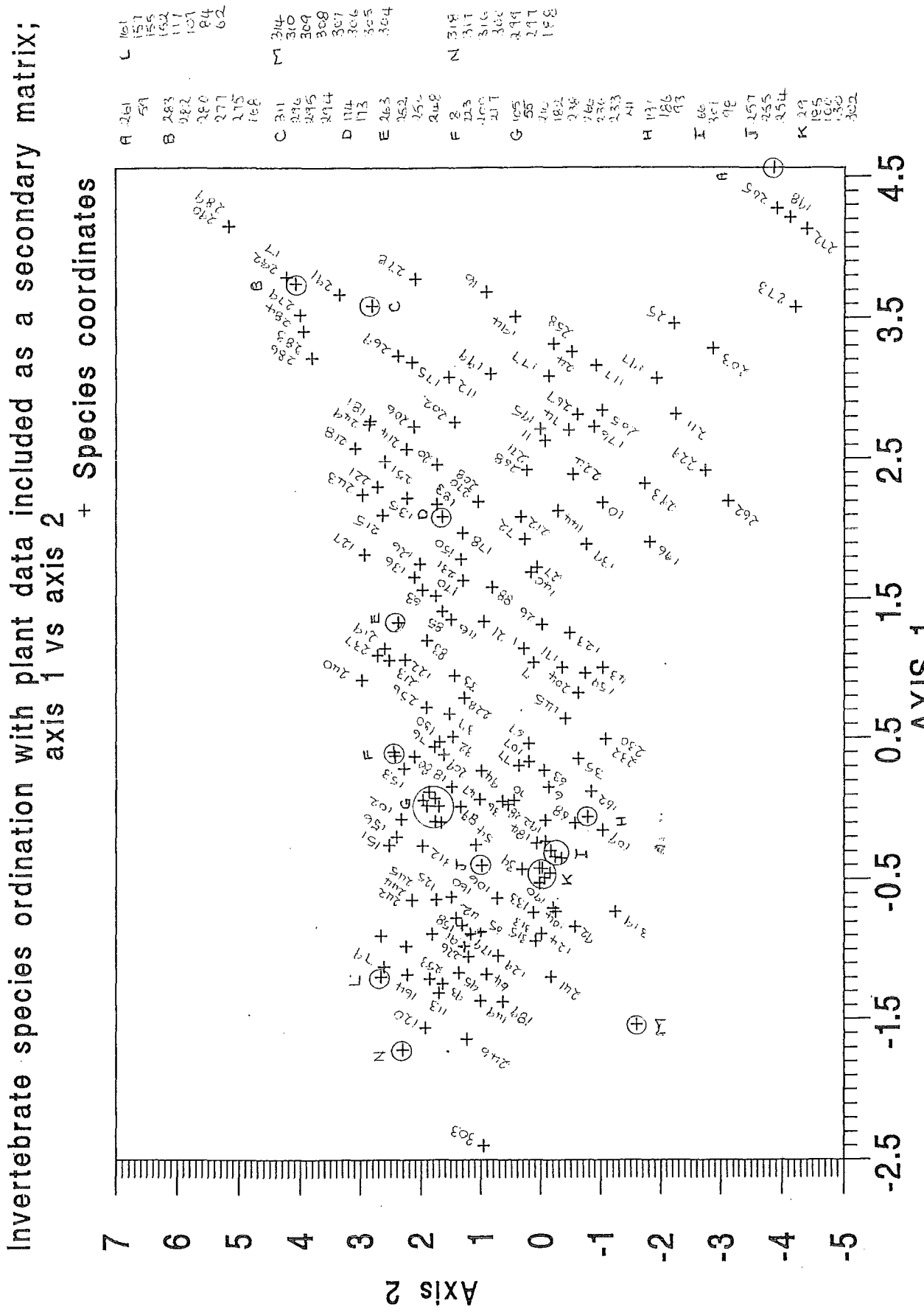
130 *Centaurium* sp
131 Thallose liverwort
133 *Lotus corniculatus*
204 *Chenopodium album*
208 *Bromus mollis*
213 *Poa compressa*
214 *Polygonum hydropiper*
215 *Verbascum virgatum*
222 *Chondropsis semiviridis*
300 *Aphanes arvensis*
302 *Solanum* sp
306 *Reseda luteola*
310 *Lachnagrostis* sp
321 *Dichelachne crinita*
324 Yellow soil lichen
327 *Mimulus moschatus*
352 Tall fescue
353 *Poa* 'blue' sp
357 *Trifolium hybridum*
400 Unidentified lichens
500 Unidentified bryophytes
600 *Solanum nigrum*
601 *Muehlenbeckia ephedroides*



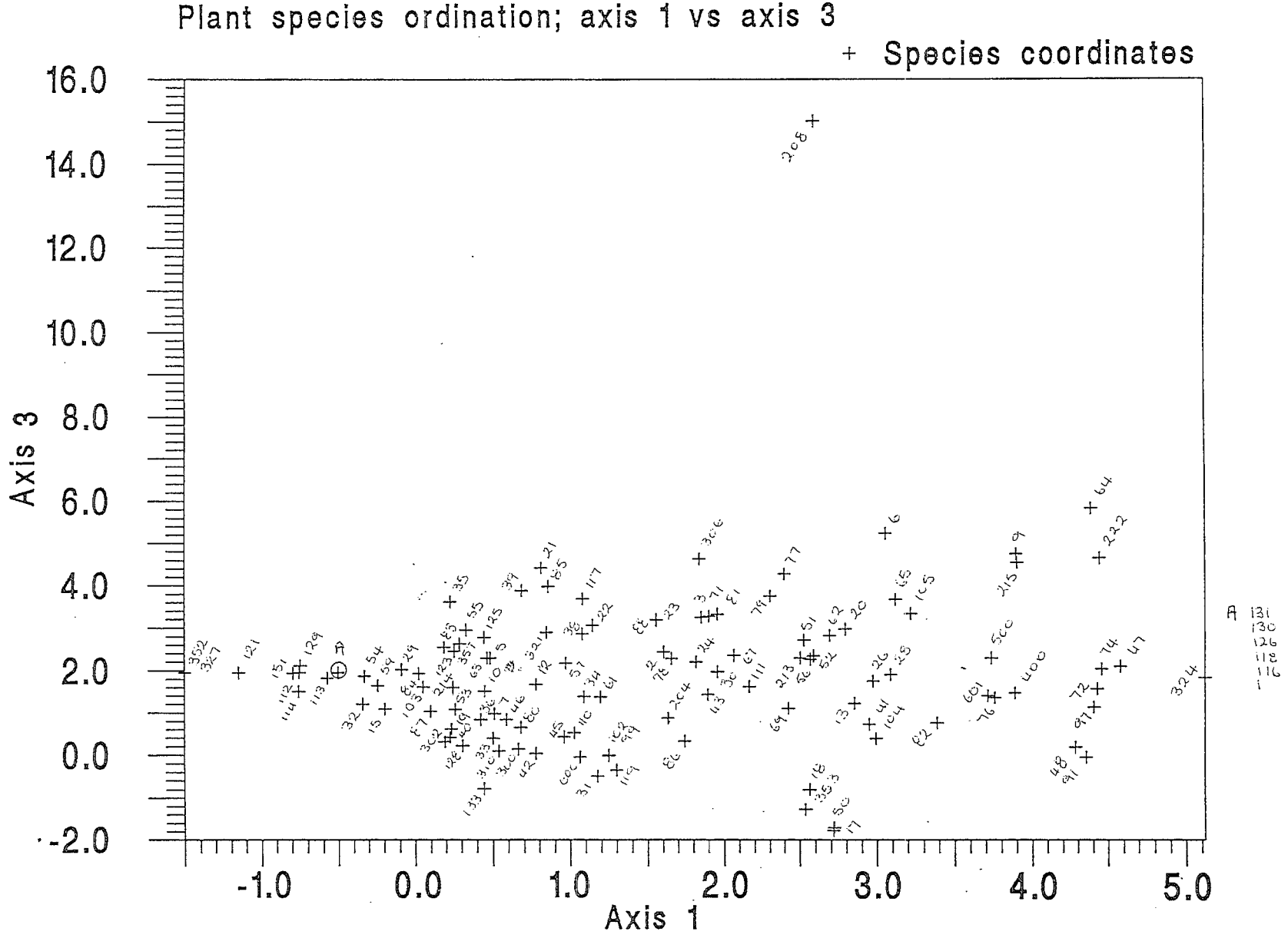
Invertebrate species ordination, axis 1 vs axis 3. Axes are in units of half change.



Invertebrate species ordination with environmental data included as a secondary matrix, axis 1 vs axis 2. Axes are in units of half change.



Invertebrate species ordination with plant species data included as a secondary matrix, axis 1 vs axis 2. Axes are in units of half change.



Plant species ordination, axis 1 vs axis 3. Axes are in units of half change.

Appendix 8

Invertebrate species lost from bulldozed habitats and their fluctuations in other habitats; - =lost from habitat, + = gained in habitat, * = still present in habitat, blank = never found in habitat in 1993. Numbers are the species Identification numbers. Location refers to whether species were found in the 1993 sample of the Experimental Shingle (S), Experimental Willow (blank) or both Experimental habitats (B).

Location	Species Id #	Ohau	Rec Trt	Cont Sh	Cont W
S	25			*	-
	41		-		
	54		-		
	55				
S	67	-		-	
B	72	*		+	-
	102	-			-
	105				
	106		+		
S	107	-	-		+
B	117			*	-
	127		+		*
	129	-	+		
	151	-			
	153	*			
	163		*		
S	171	+			
B	175			+	*
	177			-	-
S	178			-	
	182		-		
S	194			*	*
	196		-		
B	197				-
S	198			-	
B	199				*
S	203				
	206		+	*	*

	213				+
	214			-	-
	215				+
	217				
	219				*
	220				
	221			+	*
	223				
	224			-	
	233				
	236				
	237				-
	248				
	249				*
	250				
	251				+
	252				
	263				
	266				
S	278				-

Plant taxa lost from Experimental habitats, and their fluctuations in other habitats. - =lost from habitat, + = gained in habitat, * = still present in habitat, blank = never found in habitat in 1993. Numbers are the species Identification numbers (they are not in taxonomic order). Location refers to whether species were found in Experimental Shingle (S), Experimental Willow (blank) or both habitats (B).

Location	Species Id #	Ohau	Rec Trt	Cont Sh	Cont W
	13				
B	22	-	-	-	*
	24	-			+
	31	*			
B	32	*	*		
	33	-	*		
	34		-		*
	35				
	40	*	-		
S	46	*	*	+	*
	48	*		*	-
S	56			*	*
	57	*	+		
S	76				
B	80	-	-		
B	82			*	
S	91				
	99				
	102				
	103	*	*	*	
S	104	-			
S	105				
S	110	-	*	*	
	500	-	*	*	*